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PRELIMINARY ASSESSMENT OF MULTIPHASE RESTORATION EFFORTS AT LIBERTY LAKE, WASHINGTON

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16. ABSTRACT

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Liberty Lake is a 288 ha body of water located in Eastern Washington. It is a heavily utilized recreational lake when its waters and swimming beaches are not plagued by massive blooms of blue-green algae. In 1974, an alum treatment of the lake, aimed at late summer and fall release of phosphorus, successfully demonstrated the need to control internal cycling of nutrients (especially phosphorus) as well as surface and subsurface input. Macrophytes growing in rich sediments acted as nutrient pumps releasing phosphorus above the floc layer. This event--as well as flushing of the bird refuge and marshland to the south of the lake and continued input of septic tanks--overcame the alum treatment within three years. The three-year respite marked the first period in 10 years to be free of blue-green algae problems. Long-term restoration efforts began in 1978-79 with sewering of the lake periphery. Marsh runoff diversion was completed in 1979-80. Suction dredging was begun in late winter 1980, and was completed by early spring 1981. Alum treatment of the undredged area was carried out in the fall, followed by an additional treatment of the dredged area after dredging was completed. Extensive monitoring of water quality parameters began in late 1977 and has continued to assess each phase of the restoration efforts. The initial results give reason for cautious optimism and are reported in this paper.

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INTRODUCTION

While many of the intricacies of accelerated lake eutrophic processes remain poorly understood, the role and implication of excessive nutrients, especially phosphorus, has been well-elucidated (Sawyer, 1947, 1952; Ohle, 1953; Thomas, 1969; Vallentyne, 1974; Edmondson, 1972; Wetzel, 1975).

In order to deaccelerate, reverse, or at least stabilize the deterioration of a lake's water quality by overenrichment, the sources of the nutrients must be defined. In addition, the contribution of each source must be determined as accurately as possible and a mechanism set in place to divert, reduce, or mitigate that source of nutrient inflow. Furthermore, unless a concomitant educational program is established, the mitigating efforts may come to naught because a social, economic, or political decision may be made that will countermand restorative efforts. Such an action may result in the introduction of a new nutrient source or the overloading of a formerly insignificant one.

Finally, there is a need for evaluation of restorative efforts in order to predict the future of the lake in question and add to the scientific body of knowledge of lakes of similar background. This report deals with the preliminary assessment of multiphase restorative efforts at Liberty Lake, Washington.

BACKGROUND AND SEQUENCE OF EVENTS

Massive blue-green algae blooms on Liberty Lake in the early and mid-1960's provided the impetus for the Liberty Lake Homeowners Association to contact Washington State University (WSU) for advice on how to improve the water quality of their lake.

Studies by Funk *et al.* (1975,1976) which included water quality and limnological data on the lake and inflowing streams as well as observation on the nutrient recycling within the lake led to several recommendations: (1) all homes within the lake drainage area be sewered; (2) flooding and flushing of the marsh at the southern end of the lake be reduced or eliminated; (3) suction dredging and disposal of rich bottom sediment; and (4) an alum treatment to be undertaken as an initial stop-gap measure to reduce nutrient recycling within the lake.

The first major restoration effort at Liberty Lake was made in 1974 with an entire lake treatment of 95.3 metric tons of alum by WSU Environmental Engineering and Science faculty and students. This work was sponsored by the Homeowners Association and the State of Washington Department of Ecology (DOE) through the Washington Water Research Center. The procedures and treatment results are described in Funk *et al.* (1975); Funk and Gibbons (1978); and Funk *et al.* (1979).

In 1973 a sewer district was formed, three commissioners were elected, and the firm of Futrell, Redford and Saxton (now M. Kennedy Engineers) was hired to complete sewer feasibility studies. The district passed a sewer bond

issue in 1976. This group also successfully proposed a lake restoration plan in 1977 to the DOE and the U.S. Environmental Protection Agency (EPA).

The following outline shows the pattern of events as they occurred in the restoration efforts with emphasis on treatments occurring from 1978-1981. The lake and watershed also received an unscheduled 0.64 cm of volcanic ash on May 18, 1980.

(1) 1974 - Entire lake treatment with 93.5 metric tons of alum.

(2) 1978 - Construction of sewage collection system, completed in 1979.

(3) 1978 (spring) - Diversion of flood waters away from marsh. Permanent structure completed in 1979.

(4) 1980 (spring) - Mt. St. Helens ashfall.

(5) 1980 (fall) - Partial treatment (80%) of lake with alum.

(6) 1981 (spring) - Suction dredging of rich bottom sediments.

(7) 1981 (spring) - Alum treatment of dredged area completed.

PROJECT OBJECTIVES

The specific objectives of this study were to:

1. Determine as accurately as possible the prime nutrient inflow [phosphorus (P), nitrogen (N)] by surface sources into the lake.

2. Attempt to quantify septic tank seepage and groundwater flow into the lake by seepage meters or other devices.

3. Monitor changes of water quality by establishing two major lake stations and measuring standard physicochemical, biological and other limnological parameters at 1.0 to 2.0 meter (m) intervals.

4. Estimate the primary and secondary productivity and change of species, if any, as a result of restoration efforts.

5. Determine the nutrient content (P & N) of sediments and construct a sediment map to indicate where and to what depth suction dredging could achieve the best results.

6. Measure the extent of seasonal macrophyte growth and deterioration and to estimate as accurately as possible the phosphorus content of the plants.

Originally, the study plan was developed to establish baseline data one year prior to the lake restoration effort, cover the treatments and continue for one year after. Most of the actual restoration efforts were to be completed by 1979 but were delayed by a multitude of reasons to the dates shown in the outline. For these reasons, most of the data in this report represents a two year baseline and the preliminary effects of restoration on the lake. A proposal was prepared to continue studying the initial and

post-restoration conditions of the lake through January 1983. The proposal was forwarded to DOE and EPA through the Liberty Lake Sewer District and M. Kennedy Engineers and was funded. A second report is expected to be published on longer term effects by mid-1983.

STUDY AREA

Liberty Lake (Figure 1) is a soft-water lake (288 ha) of glacial origin enclosed on three sides by a small mountain range 300 to 500 m above the lake surface. Most of the watershed (3,445 ha) lies in this horseshoe-shaped basin, forested with Ponderosa pine, grand fir, Douglas fir, larch white pine, and aspen. The major tributary, Liberty Creek, originates in the higher southeastern slopes and passes through a soil series of Moscow and Springdale types before reaching the Spokane and Semihoo muck series adjacent to and in a marsh. The stream flows along the eastern margin of the marsh (and until recently, overflowed into it) before entering the lake. Most of the tributary area is underlain by quartz-feldspar-biotite paragneiss. Residential areas occupy 87% of the shoreline and overlie relatively shallow soils (Spokane series). Gneiss (west side and north shore) and Columbia River basalt (west shore) form the bedrock. A small unnamed creek enters the lake from the northwest side. Until 1978-79, waste disposal had been by septic tank and an old sewer system built in 1910, which served approximately 40% of the residents. In late 1978 and 1979, M. Kennedy Engineers completed a collection system which now serves about 2000 permanent residents. This system diverted 95% of the domestic sewage from the lake basin.

The mean residence time of lake waters is 3 years. Approximately 2.76 x 10^6 $m^3 \cdot yr^{-1}$ is lost by seepage, presumably through the bottom at the northern end of the lake. The lake may become weakly stratified for short periods of time during the mid- and late-summer period.

The lake is heavily utilized--80,000-100,000 visits per season when the swim areas and beaches are not plagued by massive blue-green algal blooms (*Gloeotrichia*, *Anabaena*, and *Aphanizomenon*).

A three-year respite from algae nuisances was made possible by the alum treatment in 1974. Details described in Funk *et al.* (1975), Funk and Gibbons (1978), and Funk *et al.* (1979).

ASSESSMENT METHODS

Earlier estimates of stream inflow, lake level, and outflow had been made from Gurley current meters and staff gauges in the lake. Precipitation was measured by standard rain and snow collector devices. From these data, Orsborn (1973) developed a water balance. Those estimates were later refined by Copp *et al.* (1976) and a nutrient budget developed for the lake by volume weighting flows with phosphorus and nitrogen data collected biweekly during the summer period and monthly during the winter period. During the low flow period of 1977, Parshall flumes were installed on the main stems of Liberty. The flumes were equipped with Manning F 3000 series flowmeters and model

Figure 1. Liberty Lake, Washington

S-4040 discrete samplers for continuous flow measurements and water sample collection. Provision was also made with Spokane Country Parks personnel for daily inspection and reading of gauges in event of equipment failure. Parks personnel were also contracted to read rain gauges and evaporation pans. Gurley meters were utilized to estimate the flow of several small intermittent streams. Storm events and runoff were similarly measured.

GROUND WATER INFLOW

Attempts to measure ground water have been made by circumscribing the lake with nests of piezometers as described by McComas (1977). Seventeen groups were installed. Each piezometer was pumped dry and allowed to refill in order to determine flow rate and insure satisfactory samples. Standard laboratory analyses for nutrients (N & P) were carried out on samples filtered through 0.45 μm pore-size membrane filters. Procedures are described in the following section on water quality.

Sample frequency has been from biweekly in the summer and fall to monthly in the winter. Vandalism problems arose when sites were made recognizable or obvious.

Seepage meters (Lee, 1977) were installed in the near shore areas to complement the piezometer nests. Vandalism occurred with a greater frequency than with the piezometers. Fifteen barrel meters were set out and only two remained undisturbed; no usable data were obtained with this method.

SEDIMENT ASSESSMENT AND NUTRIENT RELEASE STUDIES

In order to assess the contribution of nutrient release from sediments, 10 cores were taken in 1974 by Ewing piston corer. An additional 28 were taken in 1978 by a modified hand-driven piston corer. The latter device utilized a 12 x 155 cm clear plexiglass tube as both coring tube and liner. The impact of the previously used Ewing corer appeared to force the flocculent sediments away and compact the upper layers and probably resulted in lower phosphorus values when analyzed. Fourteen cores taken in 1978 were analyzed for nutrient content (phosphorus and nitrogen) and selected metals following methods outlined in the EPA Laboratory Manual for Bottom Sediments (1969) and the American Public Health Association (APHA) (1975). Four were subjected to phosphorus release tests. These core samples were augmented by 70 Ekman grab samples taken randomly across the lake to compare sediment types and construct a bottom sediment map.

Additional smaller (12 x 33 cm) cores were taken by Scuba-sampling methods, utilizing a small, stainless steel piston corer. The small corer could deposit a relatively undisturbed intact core into a laboratory test column. Both long- and short-term anaerobic, reduced oxygen, and aerobic conditions were tested against the latter cores to determine release rates of total phosphorus (TP), total soluble phosphorus (TSP), and soluble reactive phosphorus (SRP) under those prescribed conditions.

MACROPHYTE EVALUATION

Macrophyte growth was estimated by Scuba-sampling methods along 6 transects demarked by 100-500 m nylon lines laid on the lake bottom. Plants were collected at 1 m increments of depth using a round, metal sampler (0.2 m^2 area) to delineate the sample area. Three samples were collected at each depth to give a total area sample of 0.6 m^2. The macrophytes, including roots, were placed in plastic bags, tied, inflated by exhaust air from the diver's tank and allowed to float to the surface. The boat crew collected and labelled each bag.

In the laboratory, the samples were rinsed to remove sediments, drained and weighed for wet weight. Subsamples were dried for 24 hours at 100°C. The dried sample was then weighed and ashed at 500°C to obtain ash-free dry weight. Additional subsamples were digested to determine nitrogen and phosphorus content. Methods utilized were modified from Gerloff and Krombholtz (1966).

The data were plotted on a contour map and planimetered to give an estimate of the area of macrophyte growth, weight, and nutrient content.

Macrophyte measurements began each spring at ice off (March or April) and continued until late September or October, when 50% or more of the macrophytes senesce and deteriorate. First estimates were made in fall 1974. Measurements for restoration evaluation began in March 1978, and will continue through 1981.

BENTHIC INVERTEBRATES

Extensive collection of benthic invertebrates began in March 1978. Prior to that time, collections were made on the basis of monitoring the initial alum treatment of the lake. Sampling until August 1978 was carried out at 17 locations and consisted of random Ekman grab samples in each of the areas delineated by the bottom sediment map. For the past two years, a modified stratified random sampling method was adopted which takes into account the three general substrate types (Nelson, 1981). The first can be described as a silt composed of finely divided organic matter, relatively decomposed, and is characteristic of the middle portion and southern end of the lake under 6-9 m of water. The second is adjacent to and partially in the littoral zone. This area experiences heavy macrophyte growth during the spring and summer with Elodea, Potamogeton, and Ceratophyllum being the dominant plants. The sediment is muck-like in appearance and contains partially decomposed plants with some parts readily distinguishable. The third major sediment type is composed of a wide variety of inorganic particle sizes ranging from sand to gravel and rock. Overlying waters are 0 to 6 m in depth.

A random site is selected by superimposing a grid system on a map of the lake. Each grid represents a 60 x 60 m square. The grids within each substrate are numbered and a site selected by random number tables. The number of samples taken from each site depend upon the variation encountered during the previous sample period. The usual number of samples taken is 12. Sample frequency is once every three weeks during spring, summer, and fall and once per month in the winter.

Ekman grab samples are used in the open waters and a box-type sampler with a bladed screen as described by Minto (1977) is used in areas of heavy macrophyte growth. Material from the samples is initially screened in the field utilizing three screens in tandem with a high wall on the first. The screen set is composed of one each of a number 6, 20, and 28 U.S. Standard sieve sizes. The screened materials are placed in plastic bottles preserved with 40% formalin solution mixed with rose bengal dye (Mason and Yevich, 1967) to facilitate laboratory sorting. Additional screening occurs in the laboratory with U.S. Standard sieve sizes number 6, 10, 16, 25, 30, and 40. After microscope scanning, the material is placed in subsampling tray divided in seventy, 3 cm^2 units and mixed for random distribution. Subsamples for extensive identification are collected from five of the 3 cm^2 units as determined by a random number table. The data obtained from each subsample is checked for randomness by use of the chi-squared test (Elliot, 1971).

WATER QUALITY AND PRODUCTIVITY MEASUREMENTS

Standard field physicochemical measurements were made biweekly during the growing season and monthly in the winter at 1.0 m intervals top to bottom at two lake stations and on inflowing streams. Measurements included temperature, light transmission, dissolved oxygen, alkalinity, pH, conductivity and Secchi disk. Both wet chemistry and calibrated probes were utilized (light--Kahlsico Gemware; conductivity, dissolved oxygen, and pH--Hydrolab II). Sample sites are shown in Figure 2.

Laboratory Chemical Analyses

Laboratory chemical analyses of inflowing stream and lake samples were carried out, utilizing Technicon Autoanalyser (II) procedures.

Various phosphorus constituents measured were total phosphorus (TP), soluble reactive phosphorus (SRP), and total soluble reactive phosphorus (TSP). Nitrogen components were total Kjeldahl (TKN), ammonia (NH_3), nitrite (NO_2), and nitrate (NO_3). Conservative nutrients such as silica, sodium, potassium, calcium, magnesium, and iron were analyzed on a monthly basis. Metallic elements were determined by atomic absorption procedures (AAS). Background and post-treatment aluminum values were also determined by AAS. Total organic carbon (TOC) was determined by the ampule method (Oceanographic International). Sample preservation and analyses followed methods outlined by APHA (1975) and EPA (1974).

Phytoplankton and Zooplankton Analysis

Phyto- and zooplankton samples, as well as samples for laboratory analyses and chlorophyll a, were collected by rapid pump methods at the same time, location, and interval as the field measurements. Water for in situ carbon-14 productivity measurements was collected in the same manner. Pump manifold and intake funnel were clear plexiglass; pump line was clear tygon 1.3 cm (id). Pumping rate was approximately 14ℓ per minute, depending upon depth, and was calculated for each sampling date. Zooplankton samples were collected by pumping water through a 60 μm mesh, nylon plankton net and cup. Killing and preservation was by formalin, ethanol, and glycerin (Schwoerbel, 1970). Identification and counts were made according to methods outlined in

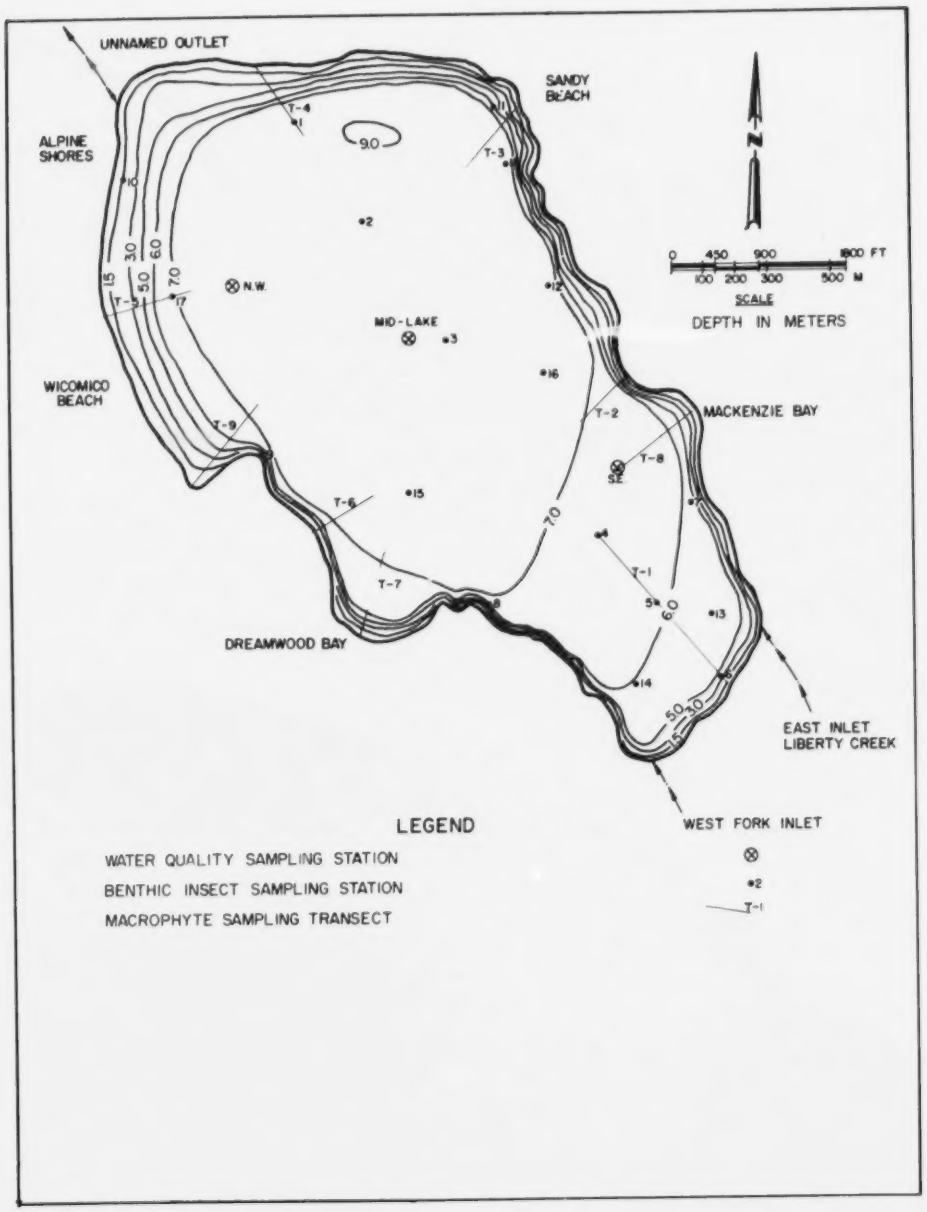


Figure 2. Water and sediment sample sites.

Edmondson (1959), Edmondson and Winberg (1971), Brooks (1957), and Pennak (1978). Successive 1 and 5 ml subsamples were examined in the laboratory for abundant and scarce individuals until 50 of the most common individuals were obtained or 20 ml of sample had been analyzed. Generally, phytoplankton counts were made upon subsamples from 1.0 ℓ unpreserved samples within 24 hours of collection. Preliminary statistical analysis involving the chi-squared test (variance to mean ratio) was employed to check for subsample homogeneity.

The remaining portion of the phytoplankton sample was preserved by modified Lugol's solution (Schwoerbel, 1970) for additional identification and measurement. If concentration was necessary, centrifugation was employed. Strip count methods were used as outlined by Edmondson (1974). Volume measurements of phytoplankton were made as described by Vollenweider et al. (1974) and Wetzel (1975). Zooplankton biomass ($\mu g \cdot m^{-3}$) was determined using the values of Hall et al. (1970), Peterka and Knutson (1970), and Bottrell et al. (1976).

Productivity Measurements

Chlorophyll a samples were fixed in the field with $MgCO_3$. Upon return to the laboratory, they were filtered through a 0.45 μm pore-size membrane and frozen. Chlorophyll a was extracted in 90% aqueous acetone solution by sonification procedures and measured by reading absorbances on a Beckman model DU 2 spectrophotometer before and after acidification with 1N HCL. Chlorophyll a and pheophytin concentration were calculated using formulas contained in Vollenweider (1974) and APHA (1975). Calibration of the spectrophotometer was checked periodically by using purified chlorophyll extract (Sigma Chemical Company).

Carbon-14 procedures were carried out in situ at both lake stations at 2.0 m intervals. The incubation period was from 10:00 to 14:00 hours. Each incubation bottle was inseminated with 5 microcuries (5 μCi) in 2 ml aliquots $NaH^{14}CO_3$. Fifty ml aliquots were filtered through 0.45 μm pore-size membrane filters and counting was done with a Nuclear Chicago-Mark II Liquid Scintilation System.

ALUMINUM MONITORING AND TREATMENT

Aluminum Background Data

Aluminum contents of inflowing streams and lake waters were monitored quarterly to semi-annually during 1977-1979. In 1980, these waters were monitored on a bimonthly to monthly basis. Analysis was by atomic absorption procedures (AAS). Tests utilizing large plexiglass tubes [20 cm (id) x 435 cm] were also made to determine whether liquid or granular aluminum sulfate ($Al_2 [SO_4]_3 \times 18 H_2O$) would be better to apply to the lake waters. Concentrations of total soluble phosphorus, total phosphorus, pH, and bicarbonate alkalinity were monitored as were settling characteristics and floc depth.

Uptake of Aluminum by Aquatic Macrophytes

Extensive sampling of aquatic macrophyte beds was carried out by Scuba-sampling procedures. Plants were dried at 105°C for 24 hours then pulverized and digested with nitric and sulfuric acids. This procedure was a modification of a wet oxidation method of plant tissue described by Jackson (1958).

Subsamples of the solubilized portions of the plant materials were analyzed by AAS.

Lake Treatment Procedures

A unique pontoon barge treatment system was devised in 1974 (Funk et al. 1975) to treat the shallow water of Liberty Lake (Figure 3). The top-mounted distribution system allows for extensive treatment into the littoral areas because of the shallow draft of the barge. In 1980, larger barges capable of carrying up to four times the amount of aluminum sulfate were employed. A grid treatment design similar to the system utilized in 1974 was employed (Figure 4). The lake was subdivided into sections and marked by flag buoys. The helmsmen and barge crews could readily determine the treatment area by buoy numbers and flag systems. The rapid formation and longevity of the floc formed easily delineated the area treated within the marked section. A new lake survey had been conducted by M. Kennedy Engineers in the spring of 1980 to accurately determine the dimensions of the lake. The recent survey allowed a more precise estimation of the amount of chemical needed for volume treatment of each section. The aluminum sulfate was delivered by semi-truck in 22 metric ton (mt) lots packaged in 36 kg bags. Fork lifts were employed to carry the pallets directly to the barges (Figures 5 and 6). In this manner, the entire 122 mt treatment was completed in less than three days. The major problem encountered as in 1974 was the irregular and slow delivery of the alum to the treatment loading area.

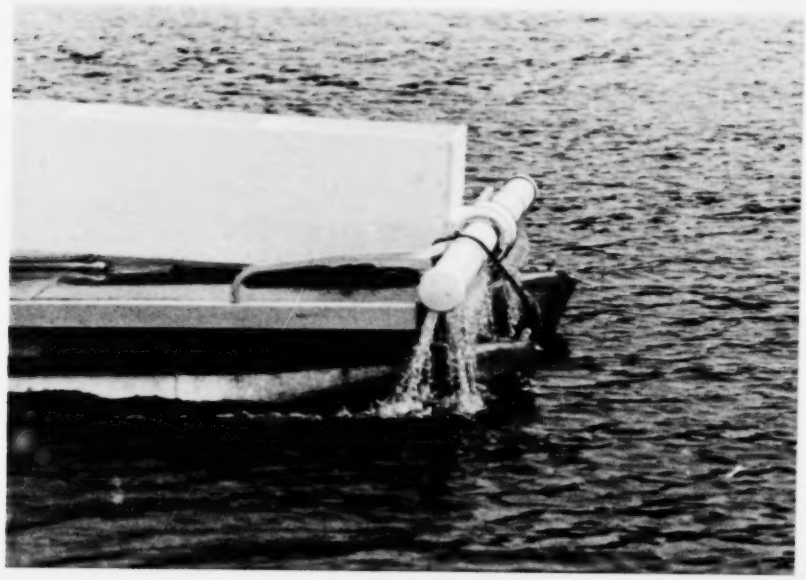
Assessment of Short-Term Treatment Effects Upon Bacteria of Significance to Water Quality

This study began in August 1980, three months before the alum treatment and employed both the membrane filter and multiple tube techniques outlined by APHA (1975). Sampling areas were the same two sites selected for the long-term assessment studies. One site (NW) was within the treatment area, the other (SE) was the control site. A series of three sample sets were taken at each site at the surface, the mid-lake depth, and the bottom. Bacteria tested for were total and fecal coliforms and fecal streptococci.

RESULTS AND DISCUSSION

WATER AND NUTRIENT BUDGET

Restoration of Liberty Lake has been a multiphase program. Preliminary efforts at reduction of nutrient flow to the lake began with a temporary diversion structure built in 1978 in cooperation with Spokane County Parks personnel at the bifurcation of Liberty Creek. The diversion allowed a large



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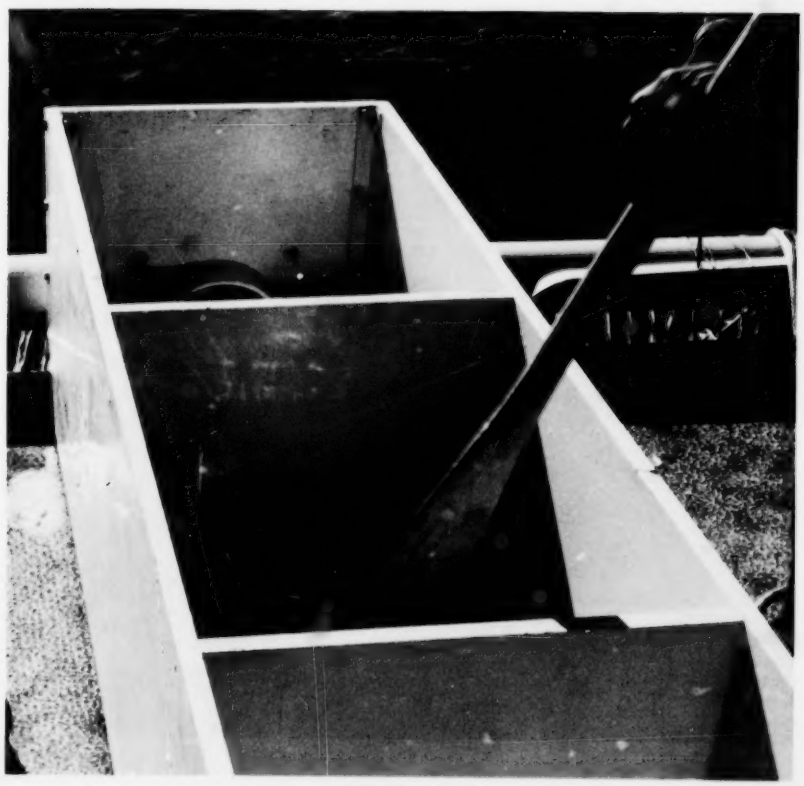


b.

Figure. 3. Treatment system. (a. side view, b. top view)

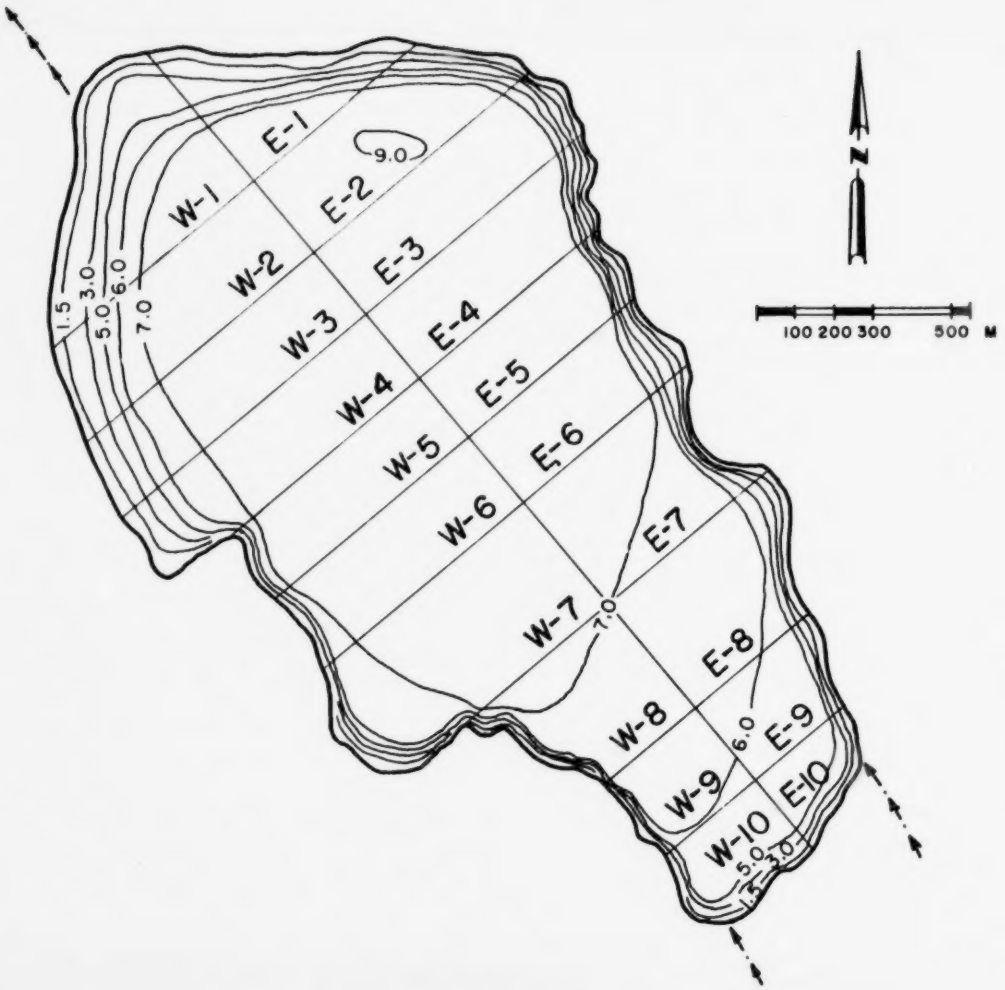


Figure 4. Grid system for treatment of Liberty Lake, Washington.

Figure 5. Large barges for 1980-81 treatment.

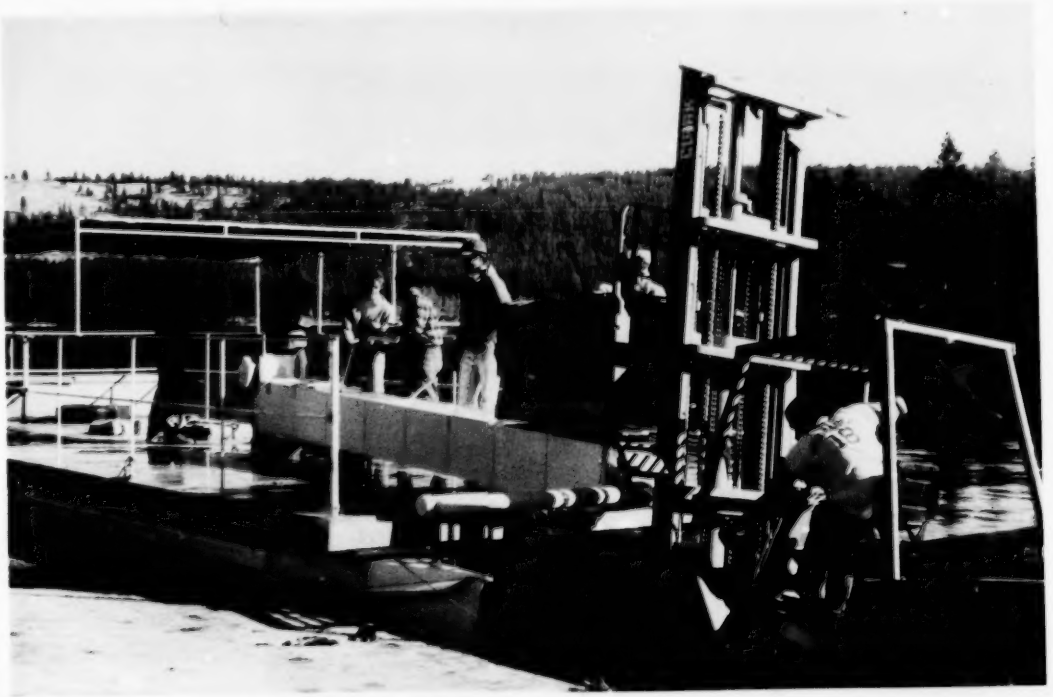


Figure 6. Loading barges by forklift.

portion of spring runoff to pass down the West Fork of Liberty Creek without flushing through the marsh to the lake (Figure 7). After the first year of operation, it was noted that with capacity stream flow in the East Fork branch nearly 30% of the water flow was still being lost to the marsh through breaks in the streambanks and overflow caused by stream bed obstructions. Even with some flushing action occurring, nutrient loading by stream overflow was reduced from approximately 148 kg P and 1137 kg N before diversion to 132.5 P and 831.6 kg N during 1978.

A permanent diversion structure, as well as stream channel repair and cleaning, was completed in 1979 by M. Kennedy Engineers. Phosphorus loading was further reduced in 1979 to 117.5 kg, but nitrogen was slightly higher (1191 kg)--possibly due to greater interaction between flowing waters and the newly cleaned stream bed. Stream hydrographs and nutrient loading are shown in Figures 8 and 9 for the East and West Fork inflows.

Water flow for 1980 was considerably greater than for 1979, but phosphorus loading was reduced to 58.7 kg. Nitrogen was reduced to 1073.6 kg for the year.

GROUND WATER MEASUREMENTS

Estimates of ground-water input made just prior to and following completion of the sewer collection system indicated that the immediate area (784 ha) around the lake contributed between 6.3×10^5 m^3 and 9.4×10^5 m^3 of water per year depending upon precipitation. This flow carried 150 to 300 kg P and 1700 to 3000 kg N toward the lake (Rector, 1979; LeCain, 1980). The remaining portion of the watershed is relatively undisturbed (2660 ha), and its nutrient addition is believed to be minor in comparison, contributing only about 12 kg as P and 60 kg as N to the lake. Table 1 shows mean values of piezometer samples for 1978-79.

TABLE 1. MEAN CONCENTRATION OF PHOSPHORUS AND NITROGEN IN THE GROUNDWATER AT LIBERTY LAKE, WASHINGTON.

Drainage	Piezometer Sample Site	Total (mg/ℓ) Phosphorus*	Total (mg/ℓ) Nitrogen*
(#1) Northwest Side	5, 12, 13, 14	0.26	2.54
(#2) North End	22, 12, 14, 15+ (#1) values	0.153	1.41
(#3) East Side	3	0.26	1.78
(#4) MacKenzie Bay	6, 7	0.065	0.54
(#5) County Park	9, 10	0.036	0.44
(#8) Dreamwood Bay	8	0.25	5.72
(#6) Main Watershed	1, 2, 28, 29, 30	0.039	0.008

* Samples filtered through 0.45 μm pore-size membrane filters before analysis.

Figure 7. Diversion of Liberty Creek, Washington.

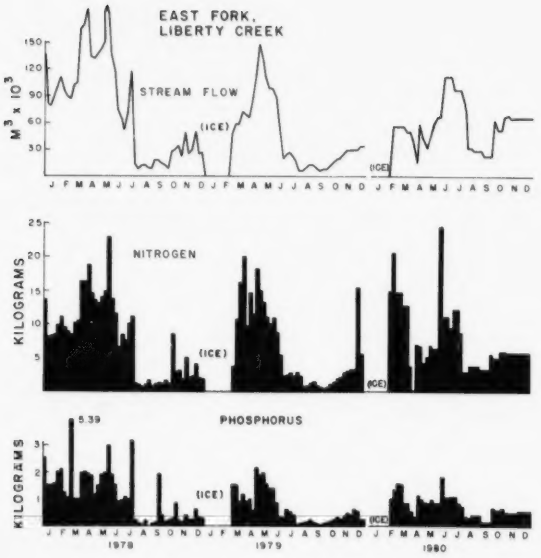


Figure 8. Hydrograph and nutrient inflow East Fork, Liberty Creek, Washington.

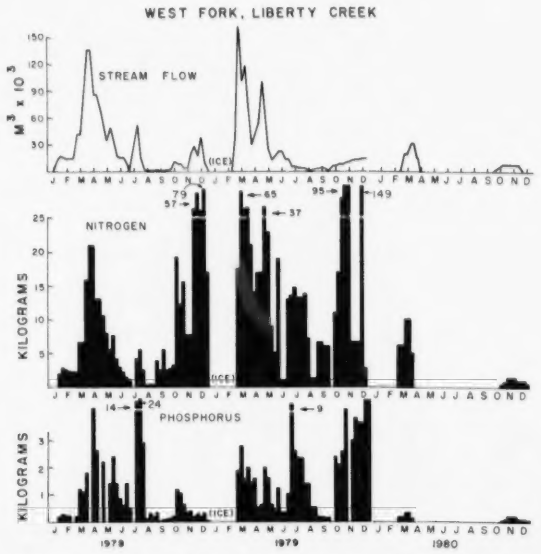


Figure 9. Hydrograph and nutrient inflow West Fork, Liberty Creek, Washington.

It is interesting to note that at Dreamwood Bay (Site #8), many of the homes have been built on fill across the natural drainage. After the collection system was completed in 1979, the piezometer samples showed sharply reduced nutrient values. It is expected that most other areas will not show such immediate improvement because of prolonged drainage bed leaching and perched water table contribution. These later phenomena were graphically demonstrated when trenches were opened for the collection system in 1979. Percolation had occurred in the sandy loam soils until a plugging of soil pores occurred or a clay lens prevented downward movement (Figure 10). The effluent then ponded and moved to the surface or laterally until it reached the soil surface on the downgrade. Considerable ground water outflow from the lake occurs in the northern portion of the lake; Orsborn (1973) estimated the flow to be 2.76×10^6 $m^3 \cdot yr^{-1}$. He utilized existing well logs to show a sand and gravel aquifer 6 m in thickness and approximately 1.5 km wide. One well (24/145-16 C1) is centrally located in the aquifer, and its elevation closely follows that of the lake. We estimate a loss of 40 to 50 kg of $P \cdot yr^{-1}$ from the lake bottom to the sand gravel aquifer based upon analysis of the well water and flow.

SEDIMENT CONTRIBUTION

Earlier investigations (Funk et al., 1975, 1979) had shown that much of the deposited material in Liberty Lake such as the gravel, clay and sand areas, release relatively few nutrients. Recent intensive laboratory work (Mawson, 1980) has verified that release of nutrients from sediments located in the southern portion of the lake could, in conjunction with macrophyte decline, support huge algal populations. Two major sediment types were tested--an refractory organic silt (ROS) representing about 70 ha and a heavy organic muck (HOM) making up 65 ha. In one test series, oxygen was added to waters over-lying the sediment column (aerobic). In another series of tests (reduced oxygen), the columns were open at the top and atmospheric oxygen was allowed to equilibrate with column water. Finally, in an anaerobic series, oxygen was removed by addition of sulfide. Conditions during tests are shown in Table 2. Summarized results for the ROS sediments are shown in Table 3; the results for the HOM sediment types in Table 4. The laboratory apparatus is shown in Figure 11.

As expected, maximum release rates occurred under anaerobic conditions for the phosphorus component measurements. Release rates from ROS sediments were also much greater, almost by a factor of 2, than the HOM sediments with the exception of SRP. Mawson (1980) believes that this may be due to more interstitial phosphorus present in the HOM as well as greater biological activity. The release of phosphorus from the ROS appears to follow a diffusion pattern, while that of the HOM is much more complex. Bottom sediment types are shown in Figure 12.

TABLE 2. AVERAGE DISSOLVED OXYGEN CONCENTRATION AVERAGE pH AND STANDARD DEVIATIONS FOR HOM AND ROS (MAWSON, 1980)

	DO (mg/l)	Standard Deviation	pH	Standard Deviation
		HOM		
Anaerobic	0.0	---	6.65	0.31
Reduced O_2	2.71	0.41	6.63	0.27
Aerobic	8.36	0.95	6.97	0.46
		ROS		
Anaerobic	0.0	---	6.55	0.16
Reduced O_2	3.1	0.53	6.70	0.15
Aerobic	6.84	0.96	7.02	0.38

HOM = Heavy Organic Muck
ROS = Refactory Organic Silt

TABLE 3. SUMMARY OF NUMBER OF OBSERVATIONS (n), SLOPE OF CONCENTRATION OVER TIME (K), CORRELATION COEFFICIENTS (r), RELEASE RATES (k*), AND CONFIDENCE LEVELS FOR AVERAGE OBSERVED CONCENTRATIONS FOR ROS SEDIMENT (MAWSON, 1980)

	n	K(mg/l-day)	r	k*(µg/m²-hr)	Confidence Levels	Analysis
Anaerobic	28	0.007	0.909	12.7	99%	T-P
Reduced O_2	30	0.001	0.330	1.22	<99%	T-P
Aerobic	16	0.000	0.092	0.186	<95%	T-P
Anaerobic	34	0.001	0.772	7.10	99%	TSP
Reduced O_2	26	0.000	0.455	1.40	95%	TSP
Aerobic	17	0.000	0.319	0.3057	<95%	TSP
Anaerobic	34	0.001	0.807	2.75	99%	SRP
Reduced O_2	26	0.000	0.277	0.393	<95%	SRP
Aerobic	17	5.283×10^{-8}	0.000	9.79×10^{-5}	<95%	SRP

TABLE 4. SUMMARY OF NUMBER OBSERVATIONS (n), SLOPE OF CONCENTRATION OVER TIME (K), CORRELATION COEFFICIENTS (r), RELEASE RATES (k*), AND CONFIDENCE LEVELS FOR AVERAGE OBSERVED CONCENTRATIONS FOR HOM SEDIMENT (MAWSON, 1980)

	n	K(mg/l-day)	r	K*(g/m²-hr)	Confidence Levels	Analysis
Anaerobic	34	0.003	0.693	5.42	99%	TP
Reduced O_2	82	0.001	0.839	2.48	99%	TP
Aerobic	26	0.000	0.411	0.399	95%	TP
Anaerobic	34	0.002	0.707	2.698	99%	TSP
Reduced O_2	82	0.001	0.779	1.41	99%	TSP
Aerobic	26	6.9×10^{-6}	0.026	0.011	<95%	TSP
Anaerobic	34	0.002	0.832	3.071	99%	SRP
Reduced O_2	82	0.001	0.750	0.10	99%	SRP
Aerobic	34	1.23×10^{-5}	0.290	1.87×10^{-2}	<95%	SRP

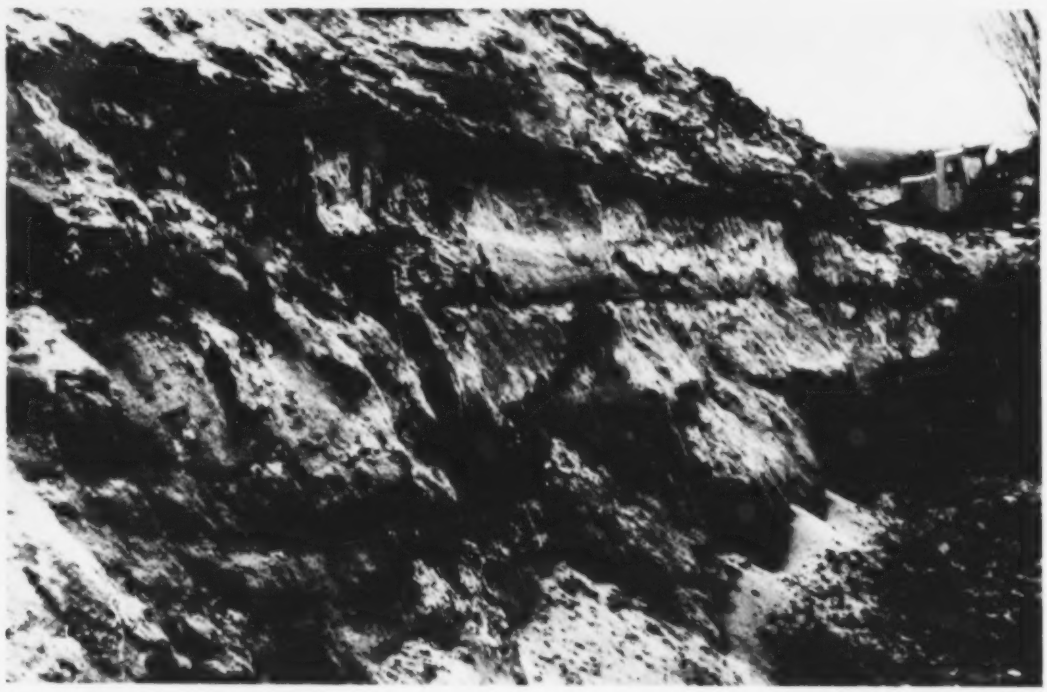


Figure 10. Soil profile of west shoreline of Liberty Lake, Washington.

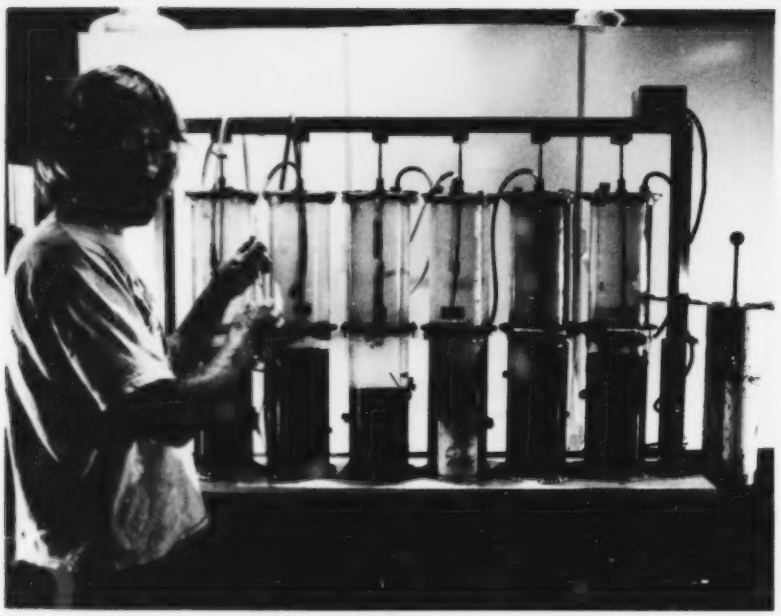


Figure 11. Laboratory apparatus for testing phosphorus release to lake water.

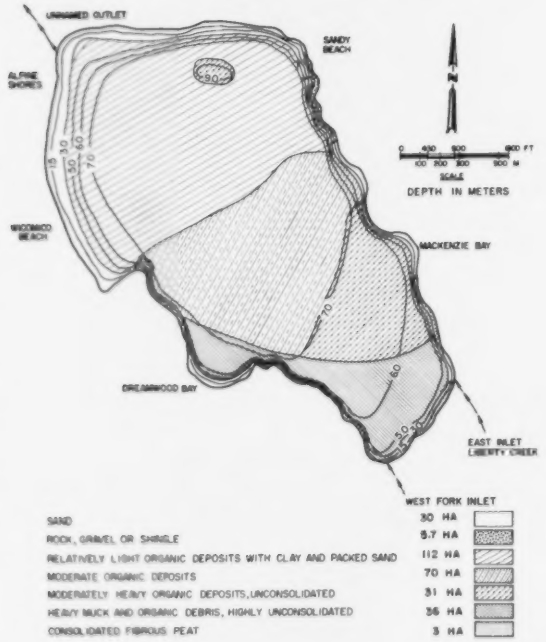

UNNAMED OUTLET
SANDY BEACH
ALPINE SHORES
WIGWAM BEACH
MACKENZIE BAY
DREAMWOOD BAY
EAST INLET LIBERTY CREEK
WEST FORK INLET
DEPTH IN METERS
SAND 30 HA
ROCK, GRAVEL OR SHINGLE 8.7 HA
RELATIVELY LIGHT ORGANIC DEPOSITS WITH CLAY AND PACKED SAND 112 HA
MODERATE ORGANIC DEPOSITS 70 HA
MODERATELY HEAVY ORGANIC DEPOSITS, UNCONSOLIDATED 31 HA
HEAVY MUCK AND ORGANIC DEBRIS, HIGHLY UNCONSOLIDATED 36 HA
CONSOLIDATED FIBROUS PEAT 3 HA

Figure 12. Bottom sediment characteristics.

MACROPHYTES

Macrophytes were collected along a series of transects shown in Figure 13. First measurements were made in October of 1974 and at that time approximately 58,000 kg (dry weight) of plants covered 80 ha of bottom area. Approximately 50% of the aquatic plants senesce in later summer and fall (September-November). In 1977 at the time of fall dieback, a moderately heavy algal bloom of Anabaena flos-aquae immediately developed. This bloom leads to increased shading of viable plants as well as the presence of blue-green algal cellular products causing additional macrophyte deterioration. These events, coupled with drought conditions and unseasonably high water temperatures, brought about increased blue-green algal growth, resulting in additional shading and by-products. Ultimately, over 75% of the macrophyte beds deteriorated and floated to the surface in large rafts. At the onset of the present investigation in 1978, maximum standing crop was about 60% of the earlier years and the previously dominant macrophyte Ceratophyllum had been replaced by Elodea canadensis especially in the southern portion of the lake. As noted by Figure 14 and a mean of four years data, the most prolific growth area is the richer more unconsolidated sediments of the southern end; 25% to 40% of the total macrophyte biomass occurs there. Tables 5, 6, and 7 show macrophyte biomass during maximum standing crop for 1978-80; biomass increased significantly during 1979. The decrease in 1980 was most likely due to increased turbidity and other effects of the Mt. St. Helens ashfall (0.64 cm) on the lake in late spring. Suction dredging in this area to remove the top 0.5 m of rich sediments also removed about 40% of the lake's prime macrophyte bed area. It is expected that this action will significantly reduce nutrient loading on the lake. Approximately 115 kg of P in plant tissue were removed without leakage of cellular content. The newly exposed sediments are poorer in nutrients and will not encourage rapid macrophyte re-growth.

TABLE 5. MACROPHYTE BIOMASS AT MAXIMUM STANDING CROP, AUGUST 2, 1978

Depth (m)	Area (m^2) x10^3	Mean AFDW* (g/m^2)	Mean AFDW (kg)	Mean Dry Weight (g/m^2)	Mean Dry Weight Contour	%P	Total P Contour (kg)
Location: Southern End and Along East Side of Lake Including McKenzie Bay							
0-1	34.37	32	1099.8	47.2	1622.3	0.19	3.1
1-2	26.41	126	3327.7	185.1	4888.5	0.22	10.8
2-3	27.82	171	4757.2	257.7	7169.2	0.16	11.5
3-4	65.24	64	4175.4	188.9	12323.8	0.11	13.6
4-5	140.68	46	6471.3	81.9	11521.7	0.16	18.4
5-6	259.97	21	5459.4	31.7	8241.0	0.17	14.0
Sub-Total Southern Portion of Lake							71.4
Location: Remainder of Lake (Eastern Portion of Lake)							
2-4	133.0	87.9	11692.0	169.0	22477.0	0.14	31.5
Total:			36982.8				102.9

* Ash Free Dry Weight

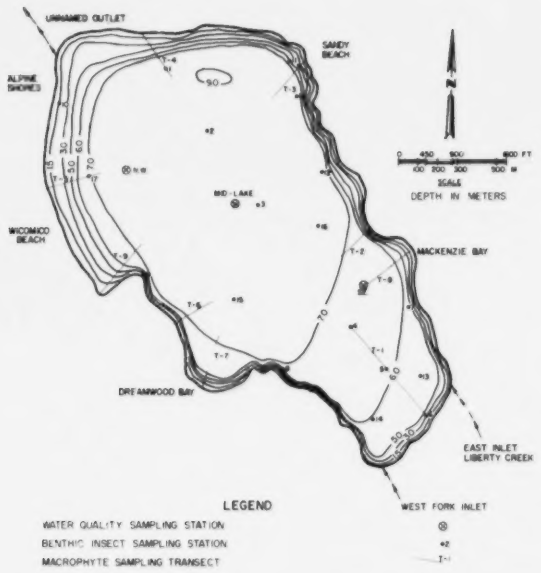


Figure 13. Water quality and benthic invertebrate sampling stations and macrophyte collection transects.

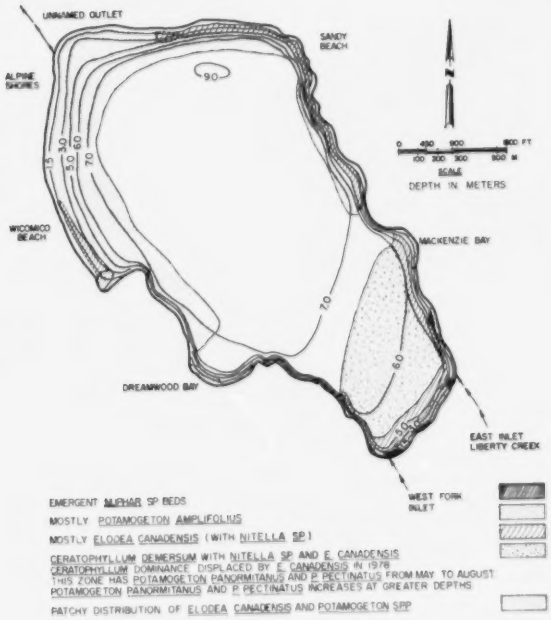

UNNAMED OUTLET
SANDY BEACH
ALPINE SHORES
WICOMICO BEACH
MACKENZIE BAY
DREAMWOOD BAY
EAST INLET LIBERTY CREEK
WEST FORK INLET
SCALE
DEPTH IN METERS
EMERGENT NUPHAR SP BEDS
MOSTLY POTAMOGETON AMPLIFOLIUS
MOSTLY ELODEA CANADENSIS (WITH NITELLA SP.)
CERATOPHYLLUM DEMERSUM WITH NITELLA SP AND E. CANADENSIS
CERATOPHYLLUM DOMINANCE DISPLACED BY E. CANADENSIS IN 1978
THIS ZONE HAS POTAMOGETON PANORMITANUS AND P. PECTINATUS FROM MAY TO AUGUST
POTAMOGETON PANORMITANUS AND P. PECTINATUS INCREASES AT GREATER DEPTHS
PATCHY DISTRIBUTION OF ELODEA CANADENSIS AND POTAMOGETON SPP

Figure 14. Distribution of aquatic macrophyte species.

TABLE 6. MACROPHYTE BIOMASS AT MAXIMUM STANDING CROP. AUGUST 21, 1979

Depth (m)	Depth (m^2) x10^3	Mean AFDW* (g/m^2)	Mean AFDW (kg)	Mean Dry Weight (g/m^2)	Mean Dry Weight Contour	%P	Total P Contour (kg)
Location:	Southern End and Along East Side of Lake Including McKenzie Bay						
0-1	34.4	33	1134.2	38	1306.0	0.19	2.5
1-2	26.4	82	2165.6	101	2667.4	0.22	5.9
2-3	27.8	133	3700.0	164	4562.5	0.16	7.3
3-4	65.2	129	8416.0	161	10503.6	0.11	11.6
4-5	140.7	103	14490.0	127	17866.3	0.16	28.6
5-6	260.0	140	36395.8	171	44459.9	0.17	75.6
6-7	86.6	85	7361.0	103	8919.8	0.22	19.6
Sub-Total Southern Portion of Lake							151.0
Location:	Remainder of Lake						
			27527.4				56.0
			101190.0				207.10

* Ash Free Dry Weight

TABLE 7. MACROPHYTE BIOMASS AT MAXIMUM STANDING CROP, AUGUST 22, 1980

Depth (m)	Area (m^2) x10^3	Mean AFDW* (g/m^2)	Mean AFDW (kg)	Mean Dry Weight (g/m^2)	Mean Dry Weight Contour	%P	Total P Contour (kg)
Location:	Southern End and Along East Side of Lake Including McKenzie Bay						
0.1	34.4	0		0			
1-2	26.4	71.6	1890.2	73.0	1927.2	0.25	4.9
2-3	27.8	143.2	3981.0	146.0	4058.8	0.30	12.2
3-4	68.2	93.0	6342.6	132.2	9016.0	0.29	26.1
4-5	140.7	118.4	16658.9	146.0	20542.2	0.27	55.5
5-6	260.0	70.3	18278.0	87.9	22854.0	0.30	68.6
Sub-Total Southern Portion of Lake							167.3
Location:	Remainder of Lake						
	Total:		11789.0				62.7
			58939.7				230.0

* Ash Free Dry Weight

BENTHIC INVERTEBRATE ASSESSMENT

In order to assess the effects of restoration efforts upon the aquatic food chains, extensive collections of benthic invertebrates began in March 1978. A stratified random sampling program was adopted for the study. The lake basin was divided into three major areas, depending upon the substrate characteristics. Each area is randomly sampled at regular intervals. The general characteristics of each of the three areas and their location in the lake basin is presented in Figure 15. A preliminary list of the taxonomic groups found within each substrate type is presented in Table 8.

The effects of dredging upon the benthos will be incorporated into the long-term assessment. Samples obtained from dredge disturbed areas indicate a limited population of invertebrates can still be found in these areas. The benthic population present is assumed to be a composite of the residual population and individuals immigrating into these areas from undisturbed areas.

The effects of the alum treatments will be evaluated on a long- and short-term basis. Data pertaining to long-term effects are still being collected at the time of this writing. Preliminary data regarding the short-term effects of the alum treatments have been obtained from samples taken immediately prior to and one week after the application of the alum. These samples are considered an indication of the immediate effects of the alum and will also be incorporated into the samples taken on a time continuum from one year prior to and one year following the application of alum.

The two sets of samples taken before and after the Fall 1980 alum treatment indicated a reduction in most benthic groups for the silt and sand/gravel substrate areas (Tables 9 and 10). The results presented are taken out of context of the overall sampling program. Conclusions regarding the immediate effect of the alum based upon the results are difficult to determine due to the few number of samples used in the silt substrate area comparison and the dissimilarity of the depths of randomly selected sites in the sand/gravel substrate.

The sampling program used for the spring 1981 alum treatment to evaluate the immediate effects of the alum treatment was designed to overcome the suspected short comings of the fall 1980 sampling. A portion of the lake basin in the Dreamwood Bay area was selected for an intensive sampling program. Fifteen sites were randomly picked in a relatively homogeneous substrate area. These sites were sampled immediately before and one week after the alum treatment. A previous study conducted in the Liberty Lake basin indicated that a piston corer-type benthic sampler provided greater estimates of the benthos than a standard Ekman grab (Nelson, 1981). As a result, a 6.8 cm (id) diameter piston corer modified from a design presented by Livingston (1955) was used in the series of samples taken at the time of the spring 1981 alum treatment (Table 11). Results based upon a two sample test (Zar, 1974) conducted upon log-transformed data indicates no significant differences at the 95% level between the population numbers of chironomids, oligochaetes, and total organisms in samples obtained before and after the alum treatment.

In other field studies concerning benthic organisms, over 100 fish stomach samples were collected during the spring and summer of 1978 with the aid of cooperating fishermen, and the Homeowners Association. Much of the data appears to be invalid because of the intensive put and take fisheries. Most of the stomachs contained corn, artificial eggs, hamburger bits and pull tabs from beverage cans rather than benthic forms. Two permit requests made by our fisheries biologist to take fish by shocking or netting were denied.

In laboratory studies, chironomid larvae (Tanytarsus dissimilus) were subjected to acute and long term alum toxicity tests (Lamb, 1980). Tests made with 240, 320, 560, and 750 $mg \cdot \ell^{-1}$ of alum did not show acute toxicity to the organism. All individuals appeared to remain fairly active and had no difficulty moving through the floc layer (1.1 mm to 3.5 mm in depth) and actively injested algal cells, Selenastrum capricornutum, supplied for food.

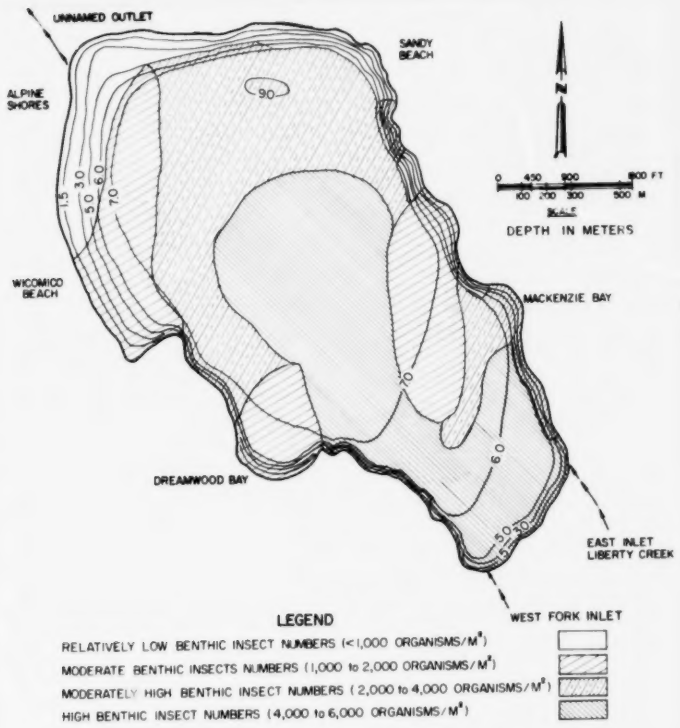


Figure 15. Benthic invertebrate distribution.

TABLE 8. MAJOR MACROBENTHIC FAUNA OF LIBERTY LAKE AND SUBSTRATE TYPE WHERE FOUND. (PRESENCE INDICATED BY X, XX DENOTES RELATIVE ABUNDANCE.)

Substrate Type	Silt	Silt/Peat	Sand/Gravel
Insecta			
Ephemeroptera			
Caenidae			
Caenis latipennis	X	X	XX
Odonata			
Libellulidae			
Tetragoneuria sp.		X	
Coenagrionidae			
Ischnua sp.		X	
Trichoptera			
Leptoceridae			
Ceraclea sp.			X
Mystacides sp.			X
Oecetis sp.			XX
Triaenodes sp.			X
Hydrospsychidae			
Cheumatopsyche sp.		X	
Hydroptilidae		X	
Diptera			
Ceratopogonidae			
Bezzia or *Palpomyia spp.*	X	XX	X
Chaoboridae			
Chaoborus sp.	XX	X	X
Chironomidae			
Ablakesmyia sp.	X	X	X
Procladius sp.	XX	XX	XX
Polypedilum sp.		X	X
Caladopelma sp.	X	X	X
Cryptochironomus sp.	X	X	X
Cryptotendipes sp.	X	X	X
Glyptotendipes sp.		X	
Dicrotendipes sp.	X	X	X
Parachironomus sp.		X	
Chironomus plumosus group	XX	X	X
Chironomus anthracinus group	X	X	
Chironomus salinarius group	X		X
Microtendipes sp.			X
Chiro 'A','B','C','D','E' spp.*	X	X	X
Stempellinella sp.			XX
Paratanytausus sp.			X
Cladotanytausus spp.	X	XX	X

Table 8. continued.

Substrate Type	Silt	Silt/Peat	Sand/Gravel
Tany 'A','B','C','D','D' spp.*	XX	XX	XX
Psectrocladius sp.	X	X	X
Corynoneura sp.		X	
Cricotopus sp.		X	X
Ortho 'A' sp.		X	X
Arachnoidea			
Hydracarnia			
Arrenuridea		X	X
Arrenurus sp.			
Pionidae			X
Forelia sp.			
Oxidae		X	
Frontipoda americana			
Limnesiidae		X	
Limnesia sp. *	X	X	X
Unidentified groups			
Crustacea			
Amphipoda			
Taltridae		X	XX
Hyalella azteca			
Gastropoda			
Physidae		X	
Physa sp.		X	
Planorbidae			
Pelecypoda *	X	X	X
Sphaeridae	X	X	X
Nematoda	XX	XX	XX
Oligochaeta			

* Additional taxonomic work required.

TABLE 9. BENTHIC SAMPLE RESULTS. TAKEN BEFORE AND AFTER ALUM TREATMENT FROM SILT SUBSTRATE. DATA ARE IN TERMS OF NUMBERS/m^2. X = MEAN, S = STANDARD DEVIATION.

		Silt Substrate	
Sampling Date:		10/19/80	11/2/80
Number of Units:		4	4
Average Depth of Units:		6.0 m	6.7 m
ORGANISM			
Insecta			
Trichoptera			
Hydroptilidae	X:	11.00	
	S:	21.50	
Leptoceridae	X:	11.00	
	S:	21.50	
Diptera			
Chaoboridae	X:	677.00	645.00
	S:	197.80	176.30
Chironomidae	X:	4,257.00	2,483.00
	S:	2,008.10	245.10
Amphipoda	X:	161.00	11.00
	S:	133.30	21.50
Pelecypoda	X:	22.00	
	S:	43.00	
Arachnoidae	X:	151.00	97.00
	S:	55.90	73.10
Oligochaeta	X:	16,609.00	11,030.00
	S:	3,319.60	2,502.60

TABLE 10. BENTHIC SAMPLE RESULTS. TAKEN BEFORE AND AFTER ALUM TREATMENT FROM SAND/GRAVEL SUBSTRATE. DATA ARE IN TERMS OF NUMBERS/m^2.
X = MEAN, S = STANDARD DEVIATION.

ORGANISM		Sand/Gravel Substrate 10/19/80	Sand/Gravel Substrate 11/2/80
Insecta			
Trichoptera			
Leptoceridae	X:	150.50	21.50
	S:	123.78	32.50
Ephemeroptera			
Caenidae	X:	10.75	
	S:	30.41	
Diptera			
Chaoboridae	X:	16.13	166.63
	S:	45.61	233.76
Chironomidae	X:	8,105.55	3,074.50
	S:	4,193.14	2,341.26
Ceratopogonidae	X:	112.88	
	S:	319.26	
Amphipoda			
Taltridae	X:	10,035.13	5,423.38
	S:	8,141.89	8,031.14
Pelecypoda			
Sphaendae	X:	5.38	129.00
	S:	15.20	254.91
Gastropoda	X:	5.38	
	S:	15.20	
Arachnoidae	X:	204.25	59.13
	S:	442.56	64.76
Olegochaeta	X:	20,091.75	8,401.13
	S:	9,108.22	8,092.29
Nematoda	X:		5.38
	S:		15.20
Total Numbers:	X:	28,737.63	17,380.63
	S:	17,450.20	16,285.49
Average Depth of Units*		2.3 m	3.7 m
* Range of Depths		2.5	1.0
		1.3	1.2
		2.0	4.1
		6.0	1.2
		1.2	6.1
		3.3	5.8
		1.0	6.2
		1.0	4.2

TABLE 11. BENTHIC SAMPLE RESULTS. TAKEN BEFORE AND AFTER THE SPRING 1981 ALUM TREATMENT FROM THE DREAMWOOD BAY AREA.
X = MEAN, S = STANDARD DEVIATION.

ORGANISM		Sampling Date 5/25/81	6/4/81
Insecta			
Diptera			
Chironomidae	X:	10,528.89	8,397.61
	S:	6,368.72	4,800.33
Chaoboridae	X:	182.16	182.16
	S:	197.75	285.96
Ceratopagonidae	X:	18.22	
	S:	70.55	
Amphipoda			
Taltridae	X:	18.22	36.43
	S:	70.55	96.14
Pelecypoda			
Sphaeridae	X:	54.65	
	S:	153.18	
Arachnoidea	X:	309.67	273.24
	S:	398.20	342.53
Oligochaeta	X:	14,445.34	11,986.17
	S:	9,524.25	4,923.61
Nematoda	X:		36.43
	S:		96.14
Total Numbers:	X:	25,557.15	20,912.04
	S:	15,772.37	9,327.92

PHYSICOCHEMICAL CONDITIONS AND LAKE WATER QUALITY

Temperature

Generally, the lake has exhibited weak thermal stratification during previous summer studies with loss of oxygen in the hypolimnion and occasional anoxic conditions (Gibbons, 1976). During this investigation, the lake did not stratify (Figure 16).

The lack of stratification may have been a reflection of the short-term climatic conditions that occurred during 1978-80. Lake water temperature varied from 0 to 25°C. Mean temperatures May through October (growth season) were 17.4°C in 1978, 18.9°C in 1979, and 17.9°C in 1980 (Figure 16). The basic effect of lack of stratification was to minimize the chemostratification of nutrients. A summary of water quality conditions is shown in Table 12

TABLE 12. SUMMARY OF 1978-79 WATER QUALITY CONDITIONS AT LIBERTY LAKE (mg/ℓ EXCEPT WHERE NOTED)

Parameter	Southeast			Northwest		
	Mean	Min	Max	Mean	Min	Max
TP	30.0	10.0	75.0	3.5	8.0	78.0
TSP	7.5	2.5	17.5	7.5	2.6	12.5
SRP	1.5	1.0	5.2	4.0	1.0	5.6
N-Ammonia	10.0	5.0	60.0	10.0	6.0	70.0
N-Nitrite-Nitrate	15.0	10.0	78.0	20.0	10.0	58.0
Total N	400.0	275.0	550.0	360.0	320.0	545.0
Alkalinity (mg/ℓ) (as $CaCO_3$)						
HCO_3	21.0	13.0	36.0	20.0	13.0	36.0
CO_3	1.0	0.0	3.0	1.0	0.0	3.0
CO_2	2.0	0.0	7.0	2.0	0.0	17.0
D.O. (mg/ℓ)	10.0	8.0	16.0	8.0	0.0	16.0
pH (-log H+)		6.5	8.5		6.0	8.6
Secchi Disc (m)	3.3	1.8	6.0	3.6	1.8	6.0
Temperature (°C)	18.2	0.0	25.0	18.2	0.0	25.0
Chlorophyll a	8.0	1.0	40.0	10.0	1.0	25.0

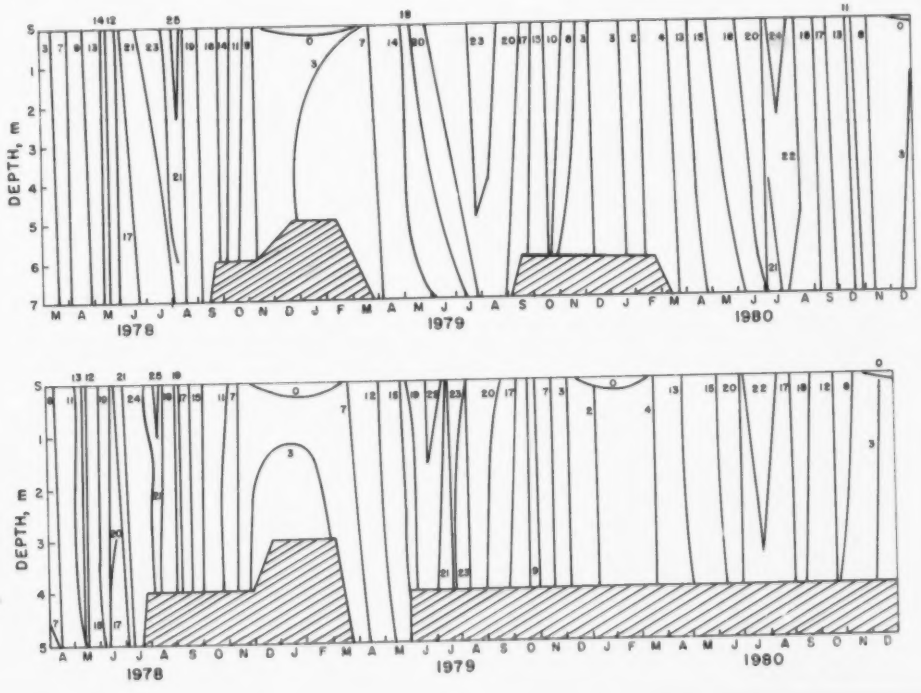


Figure 16. Temperature (°C) isopleths (1978-80) at the northwest (a) and southeast (b) stations of Liberty Lake, Washington. (Shaded areas represent bottom; water depth varied seasonally.)

Dissolved Oxygen

If thermal stratification had developed, hypolimnetic anoxia probably would have resulted in the NW section of the lake. This assumption is based upon the considerable loss of DO in the water near the bottom of that section of the lake (Figure 17) in 1978 and 1979. Dissolved oxygen measurements were taken near midday and most of the lake bottom lies in the euphotic zone. Nevertheless, it seemed that the aerobic organisms consumed oxygen at a rate higher than the ability of photosynthesizers and currents to supply it.

Dissolved oxygen in the SE section did not decrease near the bottom (Figure 17) primarily because the dense macrophytes in that area release large quantities of oxygen as a result of photosynthesis. Secondly, the SE section is 2 meters shallower than the NW section. This, coupled with the macrophyte beds, leads to a much higher level of photosynthesis and oxygen production near the bottom at the SE in comparison to the NW section of the lake.

The DO deficit in 1980 at the NW section of the lake was not as large as it was in the two previous years. The higher DO level may have been a reflection of the ash accumulation on the sediment surface (from the May 18, 1980, eruption of Mt. St. Helens), and the lack of stratification.

The ash layer may have physically retarded the diffusion of oxygen across the water sediment boundary and reduced the consumption of DO by the aerobic microbes in the sediments.

pH

The pH trends throughout the lake followed that of the primary productivity due to the unusually low alkalinity and buffering capacity of the lake. Utilization of the carbon dioxide and bicarbonate molecules in photosynthesis resulted in a rapid rise in pH. Measurements of pH ranged from 5.6 to 8.5 in the SE section and from 6.0 to 8.6 in the NW section (Figure 18). It is significant that although thermal stratification was not stable, chemostratification of hydrogen ion concentration was pronounced during the summers of 1978 and 1979 in the SW section of the lake. The difference was as great as 2 pH units between the surface and bottom waters.

Like DO, pH did not decrease as much in 1980 in the NW section of the lake as it did in 1978 and 1979--probably due to the reduction in microbial activity as the result of the ash layer deposited by the May 18, 1980 eruption of Mt. St. Helens.

Phosphorus

The concentration of phosphorus versus time has revealed no discernible seasonal pattern (Figure 19). The usually low concentration of phosphorus is probably due to its almost instantaneous uptake by algae and macrophytes. Nitrogen to phosphorus ratio was 17:1, and Schindler (1978) suggests that when N:P is more than 10:1, phosphorus is the more limiting of the two. As Wetzel (1975) has noted, the concentration of SRP and TSP are not as significant as the rate of interchange between SRP, TSP, and particulate phosphorus in

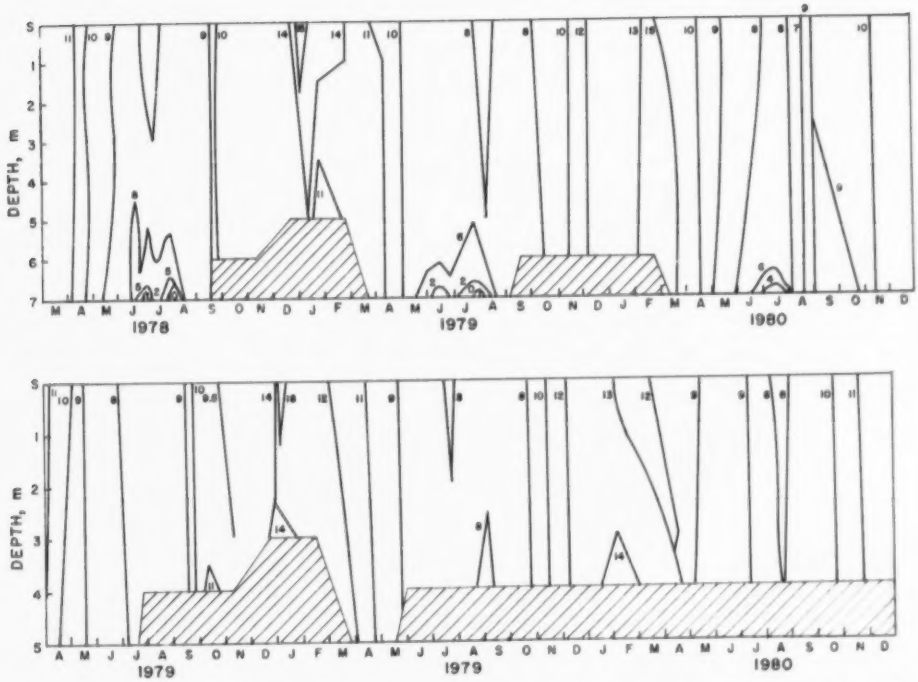


Figure 17. Dissolved oxygen ($mg \cdot \ell^{-1}$) isopleths at the northwest (a) and southeast (b) stations of Liberty Lake, Washington. (Shaded area represents bottom; water depths varied seasonally.)

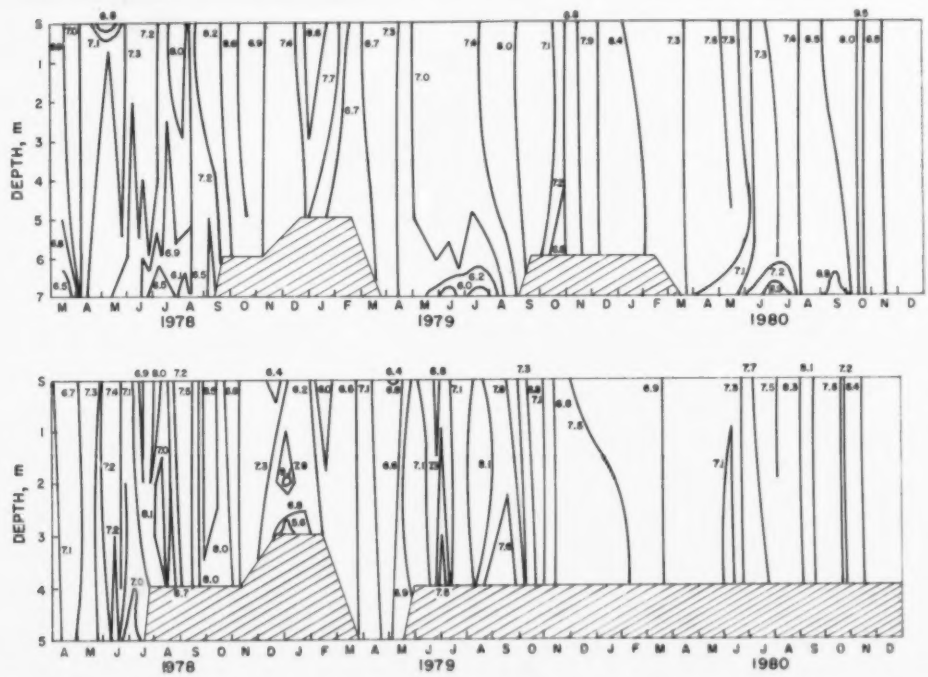


Figure 18. pH isopleths at the northwest (a) and southeast (b) stations of Liberty Lake, Washington. (Shaded area represents bottom; water depths varied seasonally.)

controlling primary production. The ability of the blue-greens to accumulate phosphorus far in excess of their immediate needs (Fogg et al. 1973; Whitton, 1973), helps to account for our repeated observations (1971-80) that masses of Anabaena flos-aquae, A. spiroides, and Gloeotrichia echinulata first appear in the vicinity of decaying macrophytes and near the bottom sediments before rising in the water column.

Conservative Elements: Calcium, Magnesium, and Potassium

Conservative elements such as calcium, magnesium, and potassium had been regularly measured to provide information on perturbations as a result of dredging or alum treatments. Variation with depth of Ca, Mg., and K ion concentration was not observed during the study due to the lack of thermal stratification in the lake. Change in concentration of these elements with time did not follow any recognizable pattern as a result of the narrow range of concentrations that were observed in the lake during 1978-80. In the southeast section of the lake, the ranges of Ca, Mg, and K concentrations were 3.4-5.4 $mg \cdot \ell^{-1}$, 0.2-1.4 $mg \cdot \ell^{-1}$, and 0.7-1.4 $mg \cdot \ell^{-1}$, respectively. In the northwest section of the lake, the ranges of Ca, Mg, and K concentrations were 3.1-8.1 $mg \cdot \ell^{-1}$, 0.2-2.1 $mg \cdot \ell^{-1}$, and 0.5-1.4 $mg \cdot \ell^{-1}$, respectively (Figures 20, 21, and 22).

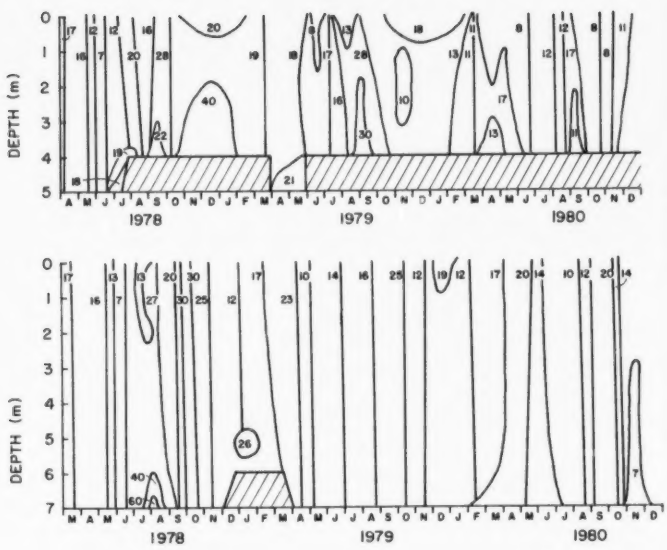
Magnesium was the only element of the three to change as a result of the alum treatment in October, 1980. The Mg concentration decreased dramatically from 1.3 to 0.2 $mg \cdot \ell^{-1}$ at the NW section of the lake and remained low through the rest of the year.

The ash fall (0.64 cm) from the May 18, 1980 eruption of Mt. St. Helens appeared to have no short-term measurable effect on Ca, Mg, or K levels in the lake.

Phytoplankton Productivity

The phytoplankton of Liberty Lake produced an estimated 8.6×10^5 kg of organic carbon and 5.9×10^5 kg of organic carbon in 1978 and 1979, respectively. The estimated annual rate of organic carbon produced in 1980 was substantially less than the two previous years (Figure 23). Three possible reasons for the reduced photosynthetic activity are: 1) reduction of nutrients entering the lake from septic tank drain fields due to completion of the sewer collection system in late 1979; 2) reduced nutrient loading as well as nutrient flux resulting from diversion of the high flow of Liberty Creek bypassing and not flushing the marsh; and 3) impeding phosphorus release from the lake sediments by the settling of ash from the Mt. St. Helens eruption. The ash covered the sediment-water interface thereby reducing phosphorus exchange performing much the same desired functions as alum floc does in reducing phosphorus in over productive lakes.

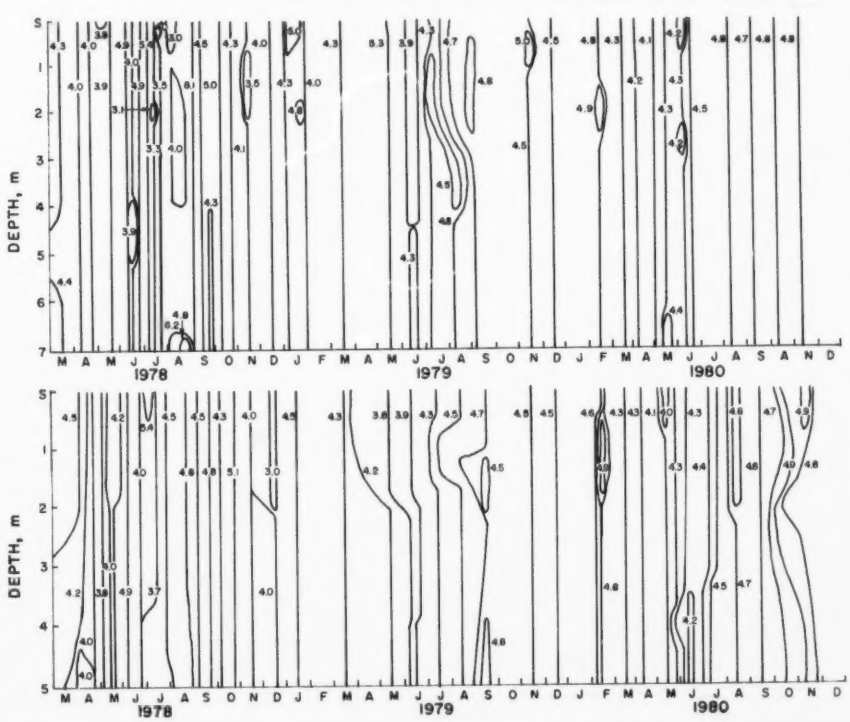
The mean rate of carbon assimilated for the growth season (May through October) was 224, 113, and 33 $mgC \cdot m^{-3} \cdot d^{-1}$ in the NW section of the lake in 1978, 1979, and 1980, respectively. For the same time periods, the rate of carbon assimilated in the SE section of the lake was 238, 181, and 38 $mgC \cdot m^{-3} \cdot d^{-1}$. The slightly higher rates of production in the SE as compared to the NW were probably due to more nutrient availability in that section than in the other.



a.

b.

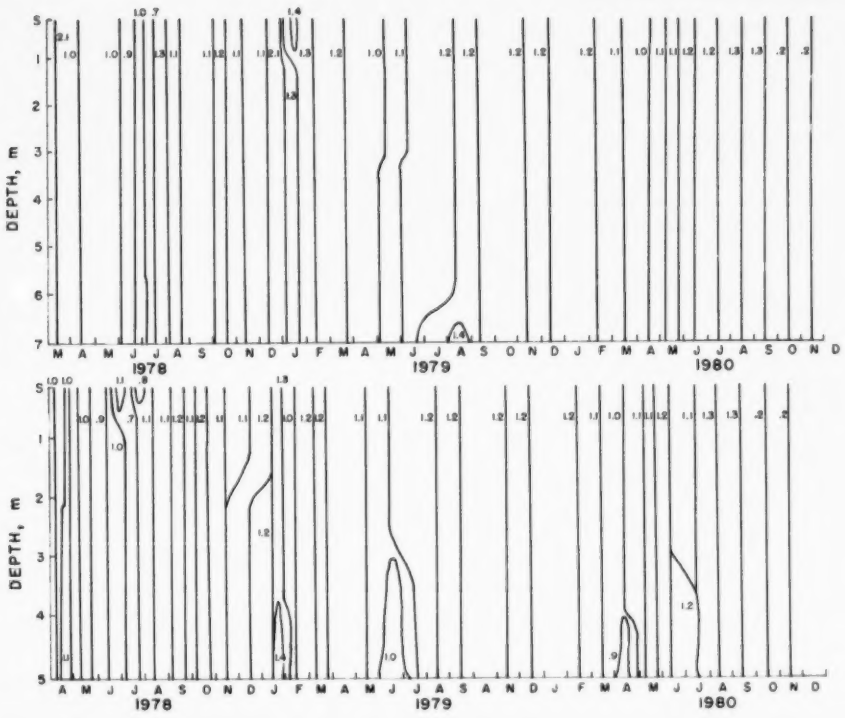
Figure 19. Total phosphorus isopleths ($\mu g \cdot \ell^{-1}$) at the southeast (a) and northwest (b) stations of Liberty Lake Washington. (Shaded area represents bottom; water depths varied seasonally.)



a.

b.

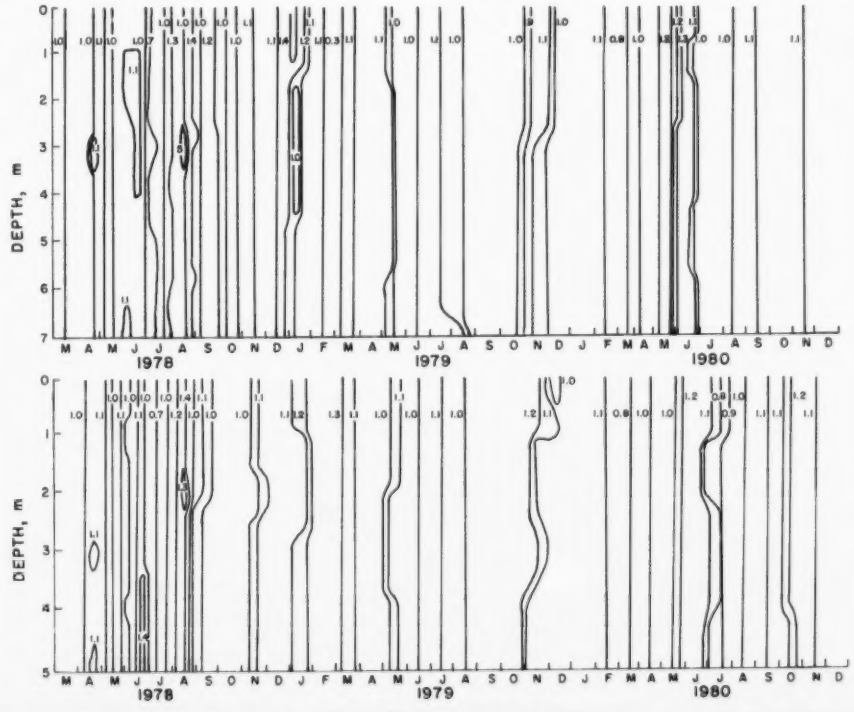
Figure 20. Calcium isopleths ($mg \cdot \ell^{-1}$) at the northwest (a) and southeast (b) stations at Liberty Lake, Washington.



a.

b.

Figure 21. Magnesium isopleths ($mg \cdot \ell^{-1}$) in the northwest (a) and southeast (b) stations at Liberty Lake, Washington.



a.

b.

Figure 22. Potassium isopleths ($mg \cdot \ell^{-1}$) in the northwest (a) and southeast (b) stations at Liberty Lake, Washington.

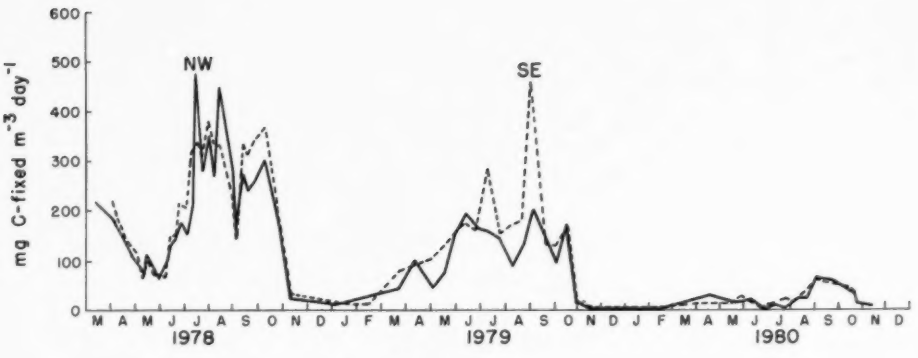


Figure 23. Mean phytoplankton production by volume ($mgC \cdot m^3 \cdot d^{-1}$) for southeast and northwest stations at Liberty Lake, Washington.

The effect of the aluminum sulfate (alum) application in October 1980, on the phytoplanktonic productivity was dramatic in both sections, although only the NW section of the lake was treated at that time. Mixing of the lake waters by wind action during the treatment caused the SE section of the lake to be partially impacted by alum. The phytoplankton productivity was reduced by 30% in the SE section of the lake and 62% in the NW section. That reduction occurred in one week. Climatic conditions during that time were similar; hence, the reduction in primary productivity was probably due entirely to the alum application. The rate of photosynthesis at the beginning of the treatment was 44 $mgC \cdot m^3 \cdot d^{-1}$ in the NW and 63 $mgC \cdot m^3 \cdot d^{-1}$ in the SE section of the lake. After the treatment (one week later), the photosynthetic rate was reduced to 16.5 $mgC \cdot m^3 \cdot d^{-1}$ in the NW and 44.5 $mgC \cdot m^3 \cdot d^{-1}$ in the SE section of the lake. Had both sections been treated with a 10 $mg \cdot \ell^{-1}$ dose of alum or had a sea-screen been in place to separate the two areas (as originally planned for the area to be dredged), the effect of the alum application would have been more pronounced.

At the beginning of the alum treatment (October 20), the blue-green algal form, Anabaena flos-aquae and the diatom, Melosira granulata were in the lake waters in profuse numbers (10,000 cells ml^{-1}). By October 24 the A. flos-aquae cells had become incorporated in the floc and had settled out of the water column. The same was true of M. granulata cells. However, by mid-November M. granulata had reestablished itself in the algae community as a dominate, but at less than 30% of its pretreatment density (from ≅2400 to <800 cells ml^{-1}). Anabaena flos-aquae did not reappear in the water after the alum treatment.

Chlorophyll a

Chlorophyll a has shown a sharp reduction over the past three years in comparison to measurements made in 1977, when lake values exceeded 30 $\mu g \cdot \ell^{-1}$ for a two-week period and ranged as high as 240 $\mu g \cdot \ell^{-1}$. This event occurred after the early decline of macrophytes and subsequent rise of Anabaena flos-aquae, A. spiroides, and Gloeotrichia echinulata blooms. Aphanizomenon

flos-aquae appeared for a short period under the ice the first week of January 1979, after a late dieoff of some of the remaining macrophytes.

The range of chlorophyll a was 0.1 to 25 $\mu g \cdot \ell^{-1}$ at the NW section and 1 to 50 $\mu g \cdot \ell^{-1}$ in the SE section of the lake. The highest chlorophyll a concentrations occurred in the fall of 1979 near the bottom of the lake in both sections. During the three year investigation, the pattern of chlorophyll a fluctuation with time was one that resulted in the annual high concentration occurring in the fall. A smaller peak followed by a dramatic decline occurred early in the spring each year. The change in phytoplankton chlorophyll a with time represents an inverse reflection of the pattern of aquatic weed production in the lake. As the aquatic weeds rapidly take up nutrients in the spring the chlorophyll a concentration decreases. When the aquatic weeds decline in the fall, the resulting increase in nutrients leads to higher levels of chlorophyll a in the water column. Chlorophyll a measurements are shown in Figure 24.

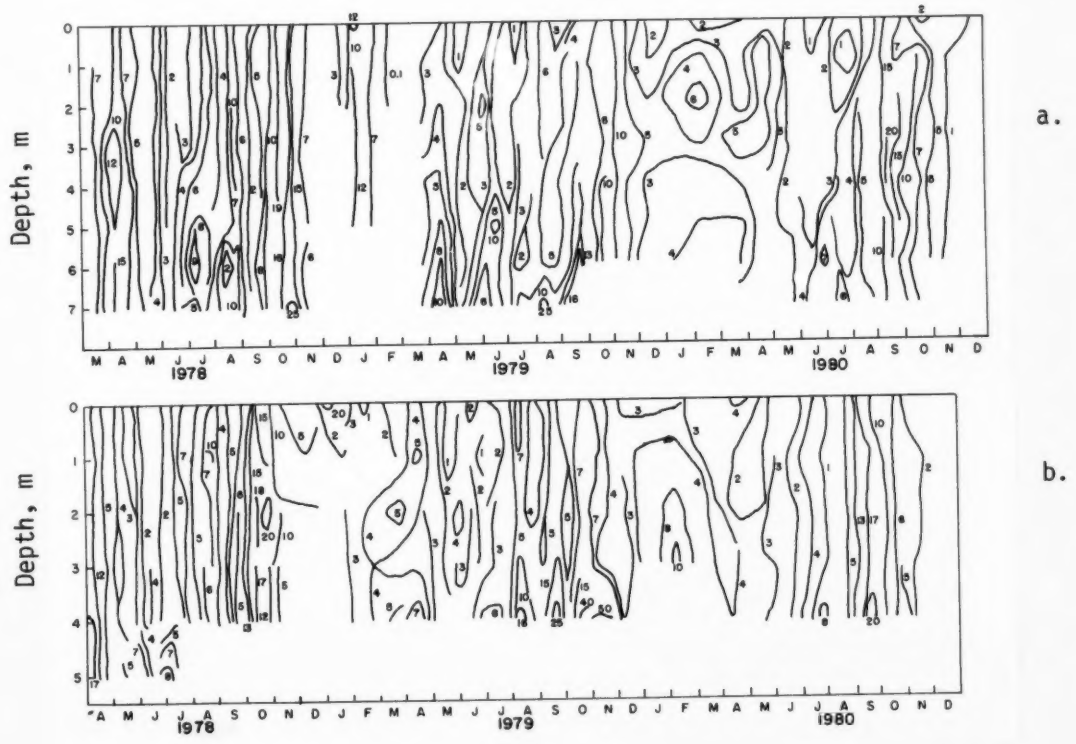

Depth, m
a.
M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
1978
1979
1980
Depth, m
b.
A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
1978
1979
1980

Figure 24. Chlorophyll a ($\mu g \cdot m^{-3}$) isopleths in the northwest (a) and southeast (b) stations at Liberty Lake, Washington.

Zooplankton

The zooplankton community of Liberty Lake, Washington was found to consist of three major groups: the Cladocera, Eucopepoda and Rotifera, and a group of lesser significance, the Diptera. A summary of principal taxa is shown in Table 13.

The Rotifers numerically dominated the community at both lake stations throughout the study period with the species Keratella cochlearis, Kellicottia longispina and Polyarthra spp making up the largest share of zooplankton density. Cladocerans, primarily Daphnia pulex and D. galeata mendotae, accounted for the largest portion of total biomass at either lake station. Copepods, represented principally by the Calanoid Diaptomus reighardi, assumed an intermediate position between the other two groups in terms of density and biomass.

Larvae of the Dipteran Chaoborus sp., the largest zooplankton encountered in Liberty Lake, formed only a minor part of the zooplankton community in both numbers and biomass.

Seasonal variations in the density and biomass of the major zooplankton groups showed similar general patterns at both lake stations. Population density peaks in the three major groups were observed to occur in the spring, with smaller peaks in the summer and the fall seasons. Except for the fall, 1979, the rotifers consistently outnumbered all other groups throughout the study. Biomass maxima were also detected in the spring, summer, and fall with the Cladocera dominating all groups in terms of dry weight standing crop.

Despite similar seasonal patterns in density and biomass among the major groups, average standing crop measures varied between the two lake stations for certain species, perhaps as a result of habitat differences. The rotifer Keratella cochlearis and smaller cladocerans like Bosmina, Diaphanizoma, and Chydorus demonstrated higher density and biomass peaks at the shallow, weedy Southeast site than at the Northwest station. In contrast, larger grazing cladocerans, especially Daphnia sp., exhibited a higher mean biomass at the deeper Northwest site, with the exception of a large peak in the fall, 1978, at the Southeast station. Average density and biomass measures for calanoid copepods were similar at both stations, although the southeast stations did reveal a higher spring peak in numbers (nauplii) and greater fall biomass maximum (adults). Cyclopoid copepods exhibited no detectable differences between stations in either standing crop measure.

The deposition of a large quantity of volcanic ash into Liberty Lake as a result of the May 18, 1980 eruption of Mt. St. Helens appeared to have an immediate effect on the zooplankton community. While zooplankton density was increasing in all three major groups prior to the eruption, the ashfall may have accelerated an early spring decline since average numbers were observed to be very low in samples collected eleven days after the event, with the exception of the rotifer group. Nevertheless, zooplankton recovery was fairly rapid since these same post-eruption samples did contain a large proportion of immature crustacean zooplankton. While the rotifer group, as well as the crustacean groups, experienced a population decline immediately following the event at the northwest site, the rotifers at the southeast site demonstrated a pronounced density maximum. At any rate, the zooplankton demonstrated normal

TABLE 13. SUMMARY OF PRINCIPAL TAXA COMPRISING THE ZOOPLANKTON COMMUNITY OF LIBERTY LAKE, WASHINGTON (9/27/78-8/22/79), INCLUDING PERCENT COMPOSITION IN TERMS OF DENSITY (No./m^3) AND DRY WEIGHT STANDING CROP ($\mu g/m^3$) OVER ALL DATES AT THE NORTHWEST AND SOUTHEAST STATIONS (GIBBONS, 1980)

Taxon	Percent Composition No./m^3		Percent Composition $\mu g/m^3$	
	NW	SE	NW	SE
Cladocera	4.1	2.8	66.5	57.7
Bosmina longirostris	*	*	0.1	0.2
Chydorus sphaericus	0.5	0.6	1.9	3.2
Ceridaphnia lacustris	*	*	0.1	0.1
Diaphanosoma brachyurum	0.1	0.3	0.6	3.0
Daphnia pulex	0.5	0.3	18.2	13.9
Daphnia schodleri	0.3	*	9.4	2.6
Daphnia galeata mendotae	1.3	0.9	19.7	21.4
Daphnia thorata	0.8	0.4	12.0	8.9
Daphnia immatures	0.5	0.3	4.2	3.9
Leptodora kindtii			0.1	0.2
Rotifera	86.3	87.2	4.3	8.0
Keratella cochlearis	24.8	41.0	1.7	4.6
Kellicottia longispina	38.3	28.2	0.7	0.9
Trichocera cylindrica	2.9	1.8	*	*
Gastropus sp.	0.6	1.0	*	*
Conchilus sp.	0.8	0.2	*	*
Polyarthra spp.	18.8	14.8	1.8	2.2
Calanoid Eucopepoda	9.3	9.8	25.5	30.1
Nauplii	7.2	8.3	2.0	3.8
Copepodids	1.6	1.1	8.6	10.0
Diaptomus reighardi	0.5	0.4	14.9	16.3
Cyclopoid Eucopepoda	0.3	0.2	2.9	3.5
Copepoidids	0.3	0.2	1.4	1.1
Mesocyclops Leuckarti	*	*	0.2	0.4
Macrocyclops albidus	*	*	1.3	1.8
Diptera	*	*	0.8	0.7
Chaoborus sp.	*	*	0.8	0.7

* = 0.1%

seasonal patterns in standing crop following the ash fall at both stations. The response of the zooplankton to the alum application phase of the restoration treatment in October 1980 was similar to that produced by the ash fall. Average density declined sharply following the treatment. However, recovery was fairly rapid, especially in the Cladoceran group which demonstrated a pronounced bloom observed only one week after the final application of alum. At the north end of the lake, an immediate reduction in numbers of each of the three orders took place. On November 4th, 10 days after the alum treatment, all three orders were recovering. By November 18th all three orders showed population decreases in keeping with the usual seasonal trends of Liberty Lake.

Zooplankton identification and counts immediately before and after the ash fall are shown in Table 14. Identification and counts just before and after alum treatment are shown in Table 15 and 16. Populations throughout most of the three-year study period are shown in Figures 25 and 26. Additional 1980 data will be published at a later date. Zooplankton evaluation will continue through 1981-82.

TABLE 14. ZOOPLANKTON DATA JUST PRIOR TO AND AFTER MT. ST. HELENS ERUPTION MAY 18, 1980

Taxon	No./m^3 4/18	5/16	5/30
Cladocera	1.126	35,210	5,732
Ceriodaphnia lacustris	0	2,062	275
Chydorus sphaericus	0	2,927	0
Daphnia pulex	0	3,057	1,457
D. galeata mendotae	0	0	525
D. schodleri	0	2,096	669
D. rosea	0	707	0
Daphnia immatures	1,126	24,361	2,806
Eucopepoda	105,886	138,739	57,005
Nauplii	96,552	69,541	28,425
Calanoid copepodids	8,851	65,046	25,688
Diaptomus bakeri	0	4,152	2,892
D. reighardi	322	0	0
Cyclopoid copepodids	161	0	0
Rotifera	543,748	330,666	102,175
Keratella cochlearis	18,989	94,767	28,575
K. cochlearis robusta	38,138	206,100	64,475
K. hiemalis	644	0	0
Kellicotia longispina	4,184	15,808	4,200
Polyarthra spp.	481,793	12,325	0
Gastropus spp.	0	1,666	4,925

TABLE 15. ZOOPLANKTON DATA DURING ALUM TREATMENT (OCTOBER 19, ONE DAY BEFORE TREATMENT: OCTOBER 24, ONE DAY AFTER COMPLETION OF TREATMENT).

Taxon	No./m^3 10/19	10/24
Cladocera	9,564	4,452
Bosmina longirostris	595	84
Chydorus sphaericus	714	28
Daphnia pulex	3,571	1,185
D. galeata mendotae	431	111
D. schodleri	1,363	161
D. thorata	331	54
Daphnia immatures	2,559	2,829
Eucopepoda	16,185	12,803
Nauplii	11,865	9,373
Calanoid copepodids	4,215	3,210
Diaptomus reighardi	105	192
Cyclopoid copepodids	0	28
Rotifera	26,305	13,975
Keratella cochlearis	17,885	8,784
K. cochlearis f. robusta	114	333
Kellicotia longispina	53	28
Trichocerca porcellus	6,857	3,831
T. cyclindrica	215	80
Polyarthra spp.	860	565
Synchaeta spp.	158	78
Monostila lunaris	163	276

TABLE 16. ZOOPLANKTON DATA BEFORE AND AFTER THE FALL 1980 ALUM TREATMENT (No./m³)

	Sample Dates				
	10/9/80	10/19/80	10/24/80	11/4/80	11/18/80
SE Lake Station					
Taxon					
Eucopepoda	56,280	31,415	57,915	51,546	39,086
Diaptomus reighardi	676	602	345	329	928
Cladocera	87,601	16,309	11,760	16,056	5,359
Daphnia pulex	2,419	5,020	6,040	4,056	1,710
D. galeata mendotae	8,065	4,778	1,744	457	175
Rotifera	163,610	76,229	47,963	49,918	30,953
Keratella cochlearis	83,865	60,928	38,915	24,889	12,949
Trichocerca porcellus	10,480	10,323	10,529	8,934	675
Polyarthra sp.	410	1,881	3,143	11,412	16,182
NW Lake Station					
Taxon					
Eucopepoda	40,571	46,124	36,223	79,926	36,858
Diaptomus reighardi	247	821	361	1,122	961
Cladocera	37,013	26,665	8,649	14,811	9,362
Daphnia pulex	5,978	11,018	4,248	9,191	5,697
D. galeata mendotae	11,582	4,965	1,337	884	499
Rotifer	65,504	78,743	40,148	46,905	23,729
Keratella cochlearis	19,578	49,466	19,320	23,597	11,575
Trichocerca porcellus	7,907	23,136	16,768	10,105	1,455
Polyarthra sp.	821	1,570	2,317	10,855	9,128

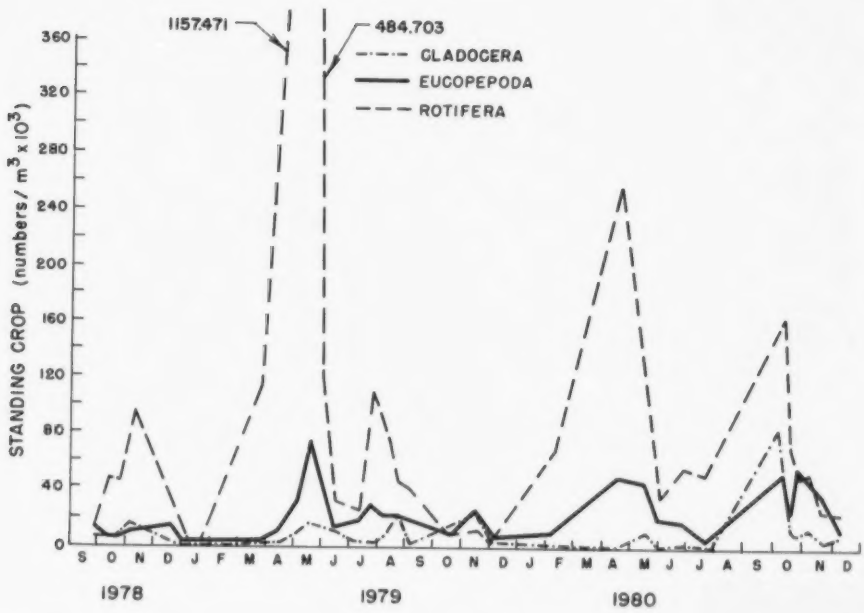


Figure 25. Mean zooplankton counts (no. x $10^3 \cdot m^{-3}$) southeast station, Liberty Lake, Washington.

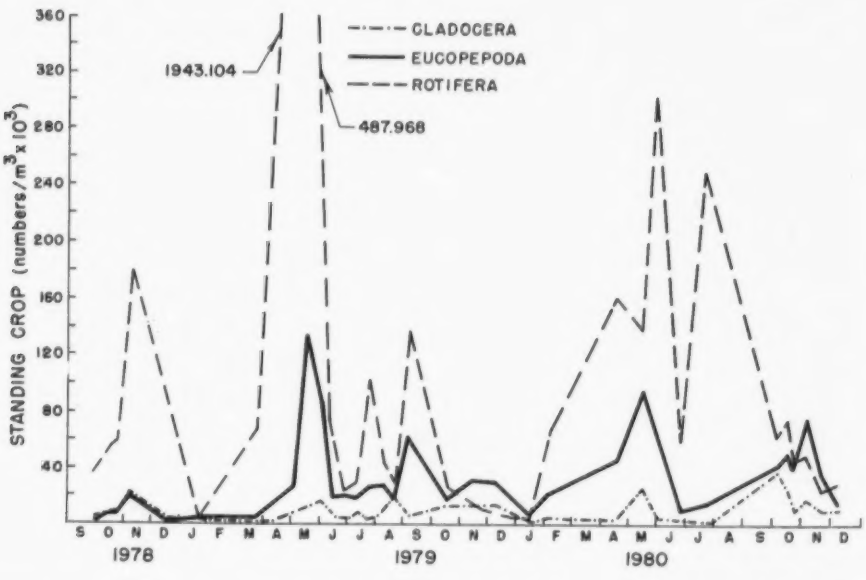


Figure 26. Mean zooplankton counts (no. x $10^3 \cdot m^{-3}$) northwest station, Liberty Lake, Washington.

ALUMINUM SULFATE TREATMENT

Laboratory Test Results

Extensive tests made in large (20 cm x 425 cm) plexiglass tubes indicated that addition of granular alum [$Al_2 (SO_4)_3 \times 18 H_2O$] directly to the Liberty Lake water gave better results than the addition of the same quantity of liquid alum. The results are shown in Table 17. Optimum dosage appeared to be 12 $mg \cdot \ell^{-1}$ dry alum and between 15 and 20 $mg \cdot \ell^{-1}$ liquid alum. Higher additions (above 20 $mg \cdot \ell^{-1}$) were not recommended because of the lack of reserve buffering at Liberty Lake.

Aluminum Background Data

Ambient levels of Al in Liberty Lake are usually between 2.0 to 4.0 $\mu g \cdot \ell^{-1}$. On occasion, such as in July and August 1980, they ranged between 2.0 and 77 $\mu g \cdot \ell^{-1}$. There is also a question as to the form of the aluminum entering and in the lake. Of the aluminum forms, gibbsite as a particle (near 0.10 µm size) is capable of passing through a 0.45 µm pore size membrane filter and is not considered to be dissolved (Hem, 1970). Streams entering the lake range from 2.0 for 1400 $\mu g \cdot \ell^{-1}$. One intermittent stream consistently high is One Way Creek, which ranges from 800 to 1400 $\mu g \cdot \ell^{-1}$. Since pH is near neutral, it is expected that the aluminum measured is present in minute particulate forms >0.40 µm.

Following the treatment on October 20, 1980 in the NW section, Al was measured at 30 $\mu g \cdot \ell^{-1}$ for a short time after the floc settled. Aluminum concentrations in the settling floc ranged from 12 $\mu g \cdot \ell^{-1}$ to 520 $\mu g \cdot \ell^{-1}$, depending upon the dispersion of the particles. By October 22 it was reduced to 12 $\mu g \cdot \ell^{-1}$ in most areas. Two days later most samples ranged from 11 to 14 $\mu g \cdot \ell^{-1}$. The untreated SE section remained at about 2 $\mu g \cdot \ell^{-1}$. A summary of selected sites is shown in Table 18.

Aluminum in Aquatic Macrophytes

The most prominent macrophyte (Elodea canadensis) that does not senesce in the fall was analyzed for aluminum content throughout 1979. Continued monitoring of this plant is planned to determine if, after two successive treatments in 1980-81, there is an increase in uptake of aluminum. Aluminum content of Elodea canadensis is shown in Table 19.

Short-Term Treatment Upon Zooplankton

The zooplankton cycles in Liberty Lake over a three-year period were discussed in the earlier zooplankton section. This section refers to the period of time just before and after the treatment. Table 15 illustrates the immediate affect. The Cladocera, Eucopepoda, and Rotifera were reduced 47, 20, and 15%, respectively, from previous numbers present.

TABLE 17. ALUMINUM SULFATE DOSAGE RESULTS

Granular Alum Dose (mg/ℓ) $Al_2(SO_4)_3 x 18H_2O$	% Reduction of Total Soluble Phosphorus				% Reduction of Total Phosphorus				pH	mg/ HCO_3	Settling Characteristics	Floc Depth cm
	Mean	Top	Mid	Bottom	Mean	Top	Mid	Bottom				
5	90	80	95	95	5	0	16	0	7.1	37	Poor	0.65
10	92	95	95	87	45.7	42	42	53	7.0	35	Good	1.3
15	95	95	95	95	60	63	63	53	6.9	26	Very Good	2.5
20	95	95	95	95	73	84	68	68	6.8	18	Very Good	3.8
Liquid Alum Dose mg/ as $Al_2(SO_4)_3$												
5	0	0	0	0	0	0	0	0	8.0	37	Very Poor	0.0
10	76.7	95	95	40	37	53	53	5	7.6	28	Poor	0.65
20	95	95	95	95	63	63	68	68	7.0	18	Good	2.5

TABLE 18. ALUMINUM PRESENT IN LIBERTY LAKE, WASHINGTON AND INFLOWING STREAMS (µg/ℓ).

Date	Lake Station (SE) S	M	B	Lake Station (NW) S	M	B	Stream Stations East Fork	West Fork	West Inlet	One Way Ck	SRO
01/05/81	2.0	2.0	2.0	2.0	2.0	2.0	54.0	59.6	18.3	1100	57.9
01/16/80							454	632	6.5	1400	2.0
02/06/80	2.4	2.0	2.0	2.0	2.0	2.0	175	300	51.8	1290	282
02/28/80	10.8	6.5	4.9	2.0	6.5	2.0	480	436	16.7	1200	
03/21/80	2.0	2.0	2.0	2.0	2.0	2.0	300	223	2.0	1290	10.1
04/03/80	2.0	2.0	2.0	2.0	2.0	2.0	340	348	2.0	1400	
04/18/80	2.0	2.0	2.0	2.0	2.0	2.0	196	136	6.5	800	10.0
05/16/80	2.0	2.0	2.0	2.0	2.0	2.0	119	260	2.0		2.9
05/29/80	2.0	2.0	2.0	2.0	2.0	2.0	130	102	2.0		9.0
06/11/80	2.0	2.0	2.0	2.4	2.0	2.0	57.5	39.2	2.0		
06/26/80	2.0	2.0	2.0	2.0	2.0	2.0	9.0	77.0	6.5		
07/09/80	2.0	2.0	2.0	2.0	2.0	2.0	12.6	4.6	2.0		
07/23/80	40.0	2.0	8.0	77.0	34.0	5.5	38.0	2.4	22.0		4.9
08/06/80	2.0	2.0	2.0	2.0	24.0	2.0	15.0		2.0		
08/20/80	2.0	2.0	2.0	2.0	2.0	2.0	23.5		2.0		
09/03/80	2.0	2.0	2.0	2.0	2.0	2.0	12.6		2.0		
10/16/80	2.9	2.0	5.0	2.0	2.0	3.4	60.0	80.0	2.0		
10/20/80	2.0	2.0	2.0	30.0	32.0	500					
10/24/80	2.0	2.0	2.0	14.2	22.0	26.0					
11/15/80	2.0	2.0	2.0	11.0	11.0	14.0	124	219	2.0		

SE - Southeast Section
NW - Northwest Section
S - Surface (6 cm)
M - Mid Depth
B - Bottom (+ 1.0 m)
SRO - Street Runoff

TABLE 19. ALUMINUM IN COMPOSITE TISSUES OF ELODEA CANADENSIS.

Date	Depth of Overlying Water	% Al Dry Weight
June 1979	2 m	0.10
August 2, 1979	2 m	0.22
August 2, 1979	2.5 m	0.19
August 2, 1979	3.0 m	0.09
December 1979	3.0 m	0.16

Treatment Effects Upon Phosphorus

Twenty-four hours after the alum treatment, Secchi disk visibility had increased to over 5 m. At this time 40 water samples were collected for phosphorus analysis at various depths, and 8 floc samples were collected from the treated area. All were analyzed for total phosphorus. Total phosphorus in the floc resting near the lake bottom ranged from 2.88 to 13.5 $mg \cdot \ell^{-1}$, that in the overlying water ranged from 0.033 to 0.037 $mg \cdot \ell^{-1}$. Soluble phosphorus was 0.001 $mg \cdot \ell^{-1}$. It is suspected that the 0.03 $mg \cdot \ell^{-1}$ level of total phosphorus measured in the upper water column was due to minute floc particles still in suspension.

Treatment Effects Upon Bacteria

A short-term study was conducted in conjunction with the alum treatment to determine the effects of the treatment upon several bacteria populations that are significant in water quality analyses. The test bacteria included fecal coliforms (FC), total coliform (TC), and fecal streptococci (FS). Sample collection and testing began three months before treatment using multi-tube procedures as outlined by Standard Methods (APHA, 1975). Sufficient data had been collected to estimate the average number of bacteria at the various depths prior to treatment.

Bacteria in the upper water column decreased very rapidly after addition of the alum. There was an instantaneous increase in bacteria numbers near the bottom. It was expected that the bacteria would be attracted to the charged aluminum colloidal particles, become immobilized, and sink. It was also thought that great numbers would become immersed in the floc and sink. In either case, the bacteria were rapidly swept from the surface waters.

Immediately following the first day of treatment, a decrease in fecal coliforms on the surface with a subsequent increase at the mid layer was noted. During the following days of treatment, this pattern continued until the surface and mid layers were reduced to less than 1 fecal coliform/100 ml. Integrating over the 6 m depth, the total number of bacteria (FC) was determined. Using this number, the settling of fecal coliform can be demonstrated visually (graph of settling), as shown in Figure 27.

A direct correlation exists (graphs) between Secchi disk and number of fecal coliform/100 ml for both the surface and bottom layers: 0.963 and 0.865, respectively. This was due to the incorporation of bacteria into the floc (the Secchi disk relating to the settling of the floc).

Similar results were observed for fecal streptococci, although they were never reduced to less than 1 fecal coliform/100 ml in the surface and mid layers. This may be due to one of two factors: 1) the influx of as many as 1000 birds (gulls) resting every night on Liberty Lake (with a low FS/FC ratio); and 2) the different absorption characteristics to the floc of gram negative rods and gram positive cocci.

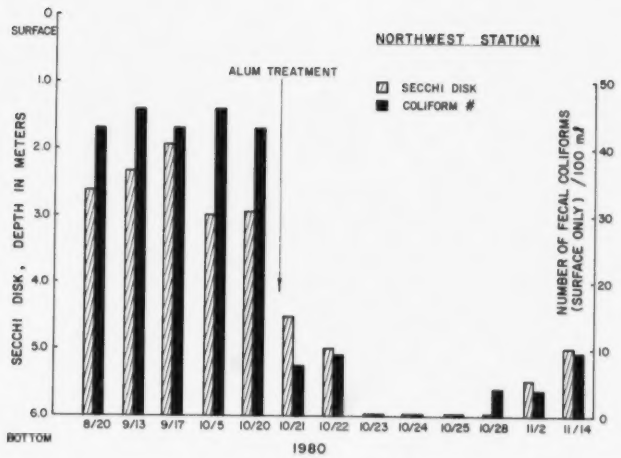


Figure 27. Effects of alum treatment upon coliform numbers at surface and upon visibility of Secchi disk at the northwest station.

SUMMARY

The low buffering capacity, shallow depth and relatively long-term detention time of Liberty Lake waters has precluded a large one-time monophase restoration effort. In addition, the high usage rate by residents, park visitors, fishermen, and water sports enthusiasts has made a stepwise treatment the most judicious route to follow. This procedure also insures protection of commercial interests whose livelihood depends upon seasonal use of the lake.

A major asset in terms of restoration protection has been the undisturbed upper watershed of Liberty Lake. Phosphorus and nitrogen content of inflowing waters is very low unless overflow flushes additional nutrients from the adjoining marsh. The marsh was formerly farm land that has been allowed to revert back to wetland through purchase by Spokane County Parks. Because of past land use practices and the large number of wild fowl attracted, it will remain a potential nutrient source.

A 15 to 20% reduction in phosphorus input has been achieved by diverting flood waters away from the marsh and to the lake through repair of the West Fork of Liberty Creek.

Sewering of 95% of the residential area around the lake has diverted 150 to 300 kg of phosphorus and 1700 to 3000 kg nitrogen from its yearly movement to the lake. It is estimated that septic tank drainage beds will continue to leach for another 4 to 7 years, depending upon the hydraulic pressure placed on them. Plugging of soil pores had occurred along the northwest area adjacent to the lake. Although deep drainage from that area had been originally away from the lake, shallow drainage of nutrient-rich waters was now directed toward the lake. In some instances, perched water tables added nutrients directly to surface drainage toward the lake.

The lake waters occasionally become weakly stratified in late July and August but the overlying layers are seldom isolated for any length of time from waters in direct contact with the sediments. Phosphorus cycling is very rapid and because of the lack of strong stratification, there is no large accumulation of phosphorus-rich phyolimnic waters. Therefore, no discernible phosphorus cycling patterns are readily apparent within the water column. However, moderately heavy algal blooms begin when the aquatic macrophytes senesce. The algal blooms are accelerated when algal cells become abundant enough to shade the remaining macrophytes and additional plants die. This pattern has been repeated many times over the past 10 years of observation and study of the lake.

The importance of the macrophyte decline in initiating algal blooms cannot be overemphasized. The 1974 and 1980 fall alum treatments were aimed at breaking that cycle. In addition, Mawson (1980) has shown that when the sediments become anaerobic for reasons such as the large scale decay of aquatic plants, a potential 150 to 280 kg phosphorus can be released. Based upon Mackenthun and Ingram (1967) estimates of phosphorus contained in algae, the potential release of phosphorus from macrophytes and sediments can theoretically produce 77 to 100 metric tons of algae in the water. After dieoff and decay of macrophyte beds and subsequent algal blooms in 1971 and 1973, we estimated over 70 metric tons of debris on the beaches alone with an

additional 4 to 5 cm of new detritus on the sediments of the SE sector of the lake.

Suction dredging in the late winter and spring of 1981 removed approximately 33% of the top sediment to a depth of 0.5 m and about 40% of the heavier macrophyte growth area. Another alum treatment of 12 $mg \cdot \ell^{-1}$ was completed after dredging to reduce phosphorus released by suspended sediments. The latter treatment also helped seal freshly exposed sediments and is expected to aid in breaking the nutrient cycle without overwhelming the limited buffering capacity of the lake.

Short-term studies to date have shown a sharp reduction in primary productivity. Chlorophyll *a* has been reduced from a high of 270 $\mu g \cdot \ell^{-1}$ in 1977 to 25 $\mu g \cdot \ell^{-1}$ at the northwest station in 1980.

Carbon 14 measurements of primary productivity has decreased steadily as shown by mean rates of 224, 113 and 33 mg $C \cdot m^{-1} \cdot d^{-1}$ for 1978, 1979 and 1980, respectively at the northwest station. For the same time period, the mean rates also decreased at the southeast station as shown by 238, 181, 38 mg $C \cdot m^{-3} \cdot d^{-1}$ for 1978, 1979 and 1980, respectively. At the same time, there does not appear to be a sharp reduction in food chain organisms except during and for several weeks following an alum treatment. The Mt. St. Helens ash fall also resulted in a short-term zooplankton reduction for about 7 to 10 days depending upon the species. A program is now in place to study post-treatment effects upon algae, zooplankton and benthic invertebrates for up to 1.5 years after the last in-lake treatment (alum).

Other short-term effects included a sharp drop in total phosphorus from 0.11 to 0.03 $mg \cdot \ell^{-1}$. Soluble phosphorus was reduced to 0.001 $mg \cdot \ell^{-1}$ for a two week period. Winds resuspended some floc particles and raised total phosphorus back to 0.07 $mg \cdot \ell^{-1}$ in surface waters. Total phosphorus in the floc ranged from 2.8 to 13.5 $mg \cdot \ell^{-1}$.

Conservative elements such as calcium and potassium were not greatly affected but magnesium was sharply reduced, possibly because of its incorporation within the photosynthetic mechanism of the algae which were then removed by the floc. Both algae and bacteria were swept from the overlying waters and the Secchi disk visibility increased from 1.5 to over 5.0 m from late October until ice cover in December.

Based upon these initial data, especially the reduction in primary productivity and coupled with the present state of good water quality conditions and healthy zooplankton and invertebrate populations, we are cautiously optimistic about the future of Liberty Lake.

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